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Using Return Intervals and Nutrient Spiraling to Examine the 2019 Nebraska Flood

Alexa Davis
University of Nebraska - Lincoln, alexadavis782@gmail.com

USING RETURN INTERVALS AND NUTRIENT SPIRALING
TO EXAMINE THE 2019 NEBRASKA FLOOD

by

Alexa Davis

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USING RETURN INTERVALS AND NUTRIENT SPIRALLING
TO EXAMINE THE 2019 NEBRASKA FLOOD

Alexa Davis, M. S.

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Advisor: Jessica Corman

In March of 2019, flooding levels in many parts of Nebraska was the worst that streams and rivers had in decades, and in some locations, on record. Theses historic floods present an optimal case study to examine how current technological resources can be used to enhance our understanding of floods and how these floods impact *in situ* stream ecosystem processes like nitrogen and phosphorus cycling. Currently, there are only a few resources available to quantify the extent of floods; for my thesis, I will focus on satellite imagery and *in situ* water level gages. Unfortunately, due to technical issues with climatic phenomena, satellite image analysis did not yield comprehensive information of flooding extent. However, by analyzing the discharge records of 94 stream gages across the state, I was able to evaluate flood stage records and calculated flood return intervals. While the flood stage was reached primarily along streams in the Niobrara, Platte, and Elkhorn River watersheds, 17 streams across the state had a flood return interval greater than 100 years. The average return interval was 48 years ± 73. The nutrient spiraling metrics in ten streams across the state were compared in the summer before and after the flood to evaluate the flood impacts on stream ecosystem processes. In 2018, the mean uptake length (S_w) of NH_4 and PO_4 was 563m ± 348 and 600 m ± 260. The vertical velocity (V_f) in 2018 for NH_4

and PO_4 was 0mm/min ± 0.42 and 0.02 mm/min ± 0.42, respectively, while the aerial uptake rate (U) mean was 0.03 ug mm/min ± 0.30 and 0 ug mm/min ± 2.61 respectively. In 2019, the mean S_w of NH_4 and PO_4 was 248 m ± 349 and 256 m ± 352. The V_f in 2019 for NH_4 and PO_4 was 0.04 mm/min ± 0.42 and 0.15 mm/min ± 0.90 while the U mean was 0.01 ± 17.16 ug mm/min and 0.57 ug mm/min ± 58.61 respectively. There were no differences ($p > 0.05$) of NH_4 or PO_4 in S_w, V_f or U between years. These data suggest that Nebraska stream ecosystems were resilient to the flood. Yet, given the high uptake lengths and low uptake velocity and areal uptake values, these data also suggest that small streams are enriched with N and P well beyond their biological demand for these nutrients. Furthering our knowledge of the quantification of floods and their impacts on stream ecosystem processes can help mitigate the deleterious consequences of floods.

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CHAPTER 1. INTRODUCTION

FLOODING

Flooding is the most frequently occurring natural disaster worldwide (Jonkman 2005) and killed at least eight million people globally in the 20th century (EM-DAT, 2004). Ninety percent of people worldwide live within 90 km of water (Kummu et al. 2011), making much of the population directly susceptible to flooding impacts. In most regions, the negative economic impacts from flooding have increased over the past few decades (Jongman et al. 2012). Within the United States, flooding is one of the most costly and frequently occurring natural disasters (Bousquin and Hychka 2019, National Flood Insurance Program and FEMA 2010).

Flooding occurs when excess precipitation cannot percolate into the ground or when water overflows onto land. Several factors such as changes in climate, land use, or flow management can impact floods and their hydrological regimes (Palmer and Ruhi 2019). Flooding can have negative and positive impacts on the surrounding environment. These impacts are dependent on many factors. The extent of damage caused by floods, biotic and abiotic, is a function of the depth, velocity and persistence of the water, as well as the dissolved and suspended load within in water (Kundzewicz et al. 2014).

2019 NEBRASKA FLOODING

In 2019, Nebraska streams and rivers had the worst flooding in decades, and in some locations the worst flooding on record (US Department of Commerce, 2020). 2019 was an exceptionally wet year in Nebraska. Every month in 2019 except February, monthly runoff in Nebraska (based on areal monthly calculation) was in the 95th percentile (USGS, 2020). In winter storm Ulmer, temperatures rose to 15.56 °C with 0.06 mm of rain in a three-day period (March 11 - March 13) (Omaha, NE Monthly Weather). With increasing flows due to excess runoff, ice in the rivers created ice jams and large slabs of ice were deposited and forced onto bridges and roadways (US Department of Commerce, 2020). Some ice chunks were estimated to weight around 3 tons (Hammel, 2019), and helped to collapse the Spencer Dam on the Niobrara River, destroying three bridges downstream of the dam.

Forty-one breaches to federal and non-federal levees were reported by the U.S. Army Corps of Engineers (Nebraska Department of Natural Resources, 2019). In total, Nebraska suffered $1.3 billion in property damages, including $449 million in damage to roads, levees, and other infrastructure, $440 million in crop losses, and $400 million in cattle from the 2019 flood (Schwarts, 2019).

DESCRIPTION OF CHAPTERS

In my thesis, I seek to answer the question of ways to quantify the flooding extent across Nebraska while also examining the flood impacts ecosystems process. Specifically, I used return intervals, climate data and nutrient spiraling to answer my questions.

In chapter 1, “Exploring the causes and impacts of the 2019 Nebraska Flood”, I quantified the flooding extent using return intervals for USGS stream gage stations and pixilated images from NASA’s NRT global mapping basic system. In evaluating some factors that influenced the flood I used two case studies of different watershed in Nebraska that were affected differently from the flood. Using climate data from the high plains' regional climate center, land use, and soil hydrological class data I was able to narrow down a climatic phenomenon called a rain on snow event. Few studies in the United States have studied flooding results from rain on snow events. I summarized the variability in examining the flood statewide and some limitations that exist for preparing for major floods.

Within chapter 3, "Examining the impacts of the 2019 Nebraska flood using Nutrient Spiraling", I compare nutrient spiraling metrics pre and post flood to examine changes in nutrient cycles in ten streams across Nebraska. Nutrient spiraling is a way to study biogeochemical cycles in streams using inputs of nutrients and a conservative tracer to evaluate changes in nutrient uptake. Although not every stream was impacted the same from the flood, I saw surprising results implicating Nebraska's high stream nutrient concentrations.

In the final chapter of the thesis, I summarize the outcomes of my study and relate both chapters to my overall research questions. I briefly discuss the broader implications of my research to highlight future research needs.

CHAPTER 2. EXPLORING THE CASES AND IMPACTS OF THE 2019 NEBRASKA FLOOD

ABSTRACT

Across the globe, 90% of people live within 10 kilometers of water sources. With the frequency, intensity, and duration of flooding and other natural disasters, increasing, society is more vulnerable than ever. In March of 2019, Nebraska had an historic flooding where over 80% of the state was in an emergency declaration. The historic floods in Nebraska present an optimal case study to examine how current technological resources can be utilized to enhance our understanding of floods and how they are impacted by climate change. Using a mix of geospatial technologies and climate data, I quantified the 2019 flood and explored some of factors that may have caused the flood. By analyzing the discharge records of 94 streams across the state, I analyzed flood stage records and calculated flood return intervals. While the flood stage was reached primarily along streams in the Niobrara, Platte, and Elkhorn River watersheds, seventeen streams across the state had a flood return interval of over 100 years. The average return interval was 48 years +/- 73. Climate weather suggests a rain on snow event influenced the accelerated ground thaw and snow melt. Throughout this research project, I identified shortfalls in the resources needed for improvement of future mitigation and planning. Furthering our knowledge and developing more resilient methods of collecting data on floods can help mitigate the deleterious consequences of floods.

INTRODUCTION

Flooding is the most frequently occurring natural disaster worldwide (Jonkman 2005) and killed at least eight million people globally in the 20th century (EM-DAT, 2004, Road and Deputy, 2005). Ninety percent of people worldwide live within 90 km of water (Kummu et al. 2011), making much of the population directly susceptible to flooding impacts. In most regions, the negative economic impacts from flooding have increased over the past few decades (B Jongman et al. 2012). Within the United States (U.S.), flooding is one of the most costly natural disasters (Bousquin and Hychka 2019), making up ninety percent of all natural disasters (National Flood Insurance Program and FEMA 2010).

FACTORS INFLUENCING FLOODS

Flooding occurs when excess precipitation cannot percolate into the ground or when streams and rivers overflow their banks. Several factors such as changes in climate, land use, or flow management can impact floods and their hydrological regimes (Palmer & Ruhi, 2019). Changes in the hydrologic regimes such as river flooding, are complex occurrences which can be caused by changes in terrestrial, socio-economic and climate systems (Kundzewicz, Hirabayashi, & Kanae, 2010). Terrestrial influences on floods include catchment size, geology, such as soil type, landscape such as agriculture, topography and soils. Land-use and land-cover changes can also enhance the potential for floods. Socio-economic factors which may impact flood potential include changes in population size and urban development. Change in the climate could physically alter many of the factors affecting floods.

NEGATIVE IMPACTS OF FLOODS ON ENVIORNMENT AND SOCIETY

Flooding can have negative and positive impacts on the surrounding environment. These impacts are dependent on many factors. The extent of damage caused by floods, biotic and abiotic, is a function of the depth, velocity and persistence of the water, as well as the dissolved and suspended load within in water (Kundzewicz et al. 2014). Water depth influences stream bank erosion and can change the streambed morphology through deposition and erosion (Dave et al. 2020). Although there can be changes in suspended sediment quantity, there can also be displacement of sediment and changes to sediment textures (Wu et al. 2015).

Floods can be damaging to the biotic regime in streams and rivers (Talbot et al. 2018), and can reduce the number of aquatic predators, thus having a cascading biotic effect on primary producers and their associated nutrient dynamics in streams and rivers (Palmer and Ruhi 2019). Reductions in primary production, organismal feeding, and reproduction have been shown due to increased suspended sediment caused by high flows (Palmer and Ruhi 2019). High flows can also influence the biogeochemical and physical properties of soil outside the floodplain ('O'Geen et al., 2010)

Abiotic conditions of the local environment determine whether organisms in streams and rivers can colonize or persist in new or changing habitat (Power et al. 1988). Floods are destructive to components of the abiotic environment such as lands, infrastructure, houses, and species communities (Tawari-Fufeyin, Paul, and

Godleads 2015). Habitat and food resources are also altered by flooding. Flash floods can influence a species' habitat selection due to the alteration of community structure such as changing mosquito oviposition (Duchet et al. 2017).

A global environmental change, affecting physical systems and ecosystems, will also affect human health in many ways (McMichael, Woodruff, and Hales 2006). Flood impacts are shown to have an uneven regional distribution (Dottori et al. 2018), meaning many of the people affected by flood will be people living in a floodplain or with lower income. People living in lesser developed regions may be more likely to leave areas affected by climate change, which may cause conflict in receiving areas (Reuveny 2007). Flashfloods from 1969-1981 were surveyed by the National Weather Service (NWS) to link mortality and the circumstances of deaths. More than twice as many deaths were associated with inadequate warnings than with those with warnings considered adequate (French et al. 1983). Duration, repetition, and intensity of floods and other natural disasters are putting the quality and abundance of our natural resources in peril.

Disease outbreaks, displacement of populations, economic losses, psychosocial disorders, injuries, and deaths are social economic problems associated with flooding (Oriaifo et al. 2020). The occurrence of natural disasters is associated with substantial costs, in the form of human and material losses or disruption of economic activity (Daniel, Florax, and Rietveld 2009).

POSITIVE IMPACTS OF FLOODS ON ENVIORMENT AND SOCIETY

While flooding is generally associated with negative societal impacts such as damages and loss of life and infrastructure, floods may provide many benefits to the environment and society (Talbot et al. 2018). Some of these benefits include recharging water sources, increasing fish production, and creating wildlife habitats (Galat et al. 1998; Talbot et al. 2018). For example rivers are dependent on floods to create unique habitats and support biological productivity and biodiversity (Talbot et al. 2018). As water levels fluctuate, habitats can expand and contract, creating shifts in resource availabilities, all while changing flow regimes alter other physical gradients (Power et al. 1988). Seasonal fluctuations in discharge are crucial to the life cycle of many riverine species (Welcomme 1985). For example, intermittent flood disturbances are important for the establishment and maintenance of the diverse groupings of vegetation communities across riverine landscapes (Munes et al. 2015).

A rich ecosystem can provide supplementary livelihoods from tourism and recreation such as fishing and hunting. Talbot and others found that small floods were important for enhancing access to food, creating recreation, and being beneficial for water regulation (Talbot et al. 2018).

WEATHER WHIPLASH- RAIN ON SNOW EVENT

Weather whiplash is a term to describe wild and rapid shifts in weather conditions (Casson et al. 2019). This typically involves two or three extremes which have a synergist effect on the surrounding environment, such as in creased precipitation leading to flooding occurring after a drought. Although recent research has only focused on warmer weather whiplash- this extreme event can occurring in winter. “Winter weather whiplash” events, are characterized by weather conditions swinging from frozen to unfrozen, or vice versa (Casson et al.

2019). Considering the lack of understanding in winter weather whiplash I defined a case study in Nebraska where a rain on snow (ROS) event lead to a winter whiplash.

A significant impact of ROS is the potential to generate high streamflow (Singh et al. 1997), while also increasing stream acidity (Eimers *et al*., 2007). The ROS event itself is typically short lived, but the effects on the ecosystems and soil temperatures may last for years (Rennert et al. 2009).

In temperate climates in spring and winter, the precipitation that falls is usually in the form of snow. Increasing temperatures allow the snow to melt, releasing the once frozen water. When this precipitation falls on snow as rain, the accumulation of water at the soil surface can create a ROS event.

ROS driven snowmelt can release latent heat into the soil horizon, resulting in an accelerated thawing of frozen ground (Putkonen and Roe 2003). The potential for ROS events to generate floods is related to the contributing area, intensity and duration of rainfall, opportunity for snow melt and the timing of the snow melt (Kattelmann 1997).

ROS events can substantially increase sub-snowpack soil temperatures (Putkonen and Roe 2003). When snow surface layers reach 0 °C, snowmelt will continue to occur whenever the cold content of the snowpack exceeds 0 °C or until the snow has completely melted (Dingman, 2015). If rainfall on the snow cover occurs, the snow melt process can be accelerated which can lead to flooding (Sui and Koehler 2001).

ROS events are relatively rare outside of a few regional areas including western Eurasia, the higher elevations of western North America, the northeastern United States, and southeastern Canada (Cohen, Ye, and Jones 2015).

In the U.S., ROS events are most occur most frequently in October through May on the coasts and in early summer and fall within the interior western United States (McCabe, Clark, and Hay 2007). The various magnitudes of ROS events have different impacts on ecosystems and the soil thermal field (Rennert et al. 2009). ROS floods are a unique hydro-meteorological phenomenon for which their severity depends on the magnitude of precipitation, the elevation of the freezing level and the water equivalent and areal extent of the antecedent snowpack (McCabe, Clark, and Hay 2007). ROS are very infrequent, occurring a few times throughout the year, but can have a large impact when they do happen (Cohen, Ye, and Jones 2015). Changes in ROS frequency is determined by two major variables: rainfall events and ground snow cover days (Ye, Yange, and Robinson 2008). Watershed recharged by snowmelt, specifically buy ROS, and is an important process that scientists have been concerned maybe affected by climate change.

The frequency of ROS events are likely to change in the future as the types and timing of precipitation change (Surfleet and Tullos 2013). Trends for ROS event frequencies are correlated with high elevation and ENSO (McCabe, Clark, and Hay 2007). Air temperature increases are associated with the decline of ROS events in the western United States, particularly at low-elevation sites (McCabe, Clark, and Hay 2007). With the effects of climate change, an increase in winter temperatures may result increase the number of ROS events (McCabe, Clark, and Hay 2007; Ye, Yange, and Robinson 2008; Leung et al. 2004).

Studying ROS events is challenging due to the difficulty of both measuring rain and snow and representing ROS events in numerical weather predictions and climate models (Rennert et al. 2009). As climate change projections can include increase temperatures, precipitation amount and duration, ROS events may increase in the future (Casson, Eimers, and Buttle 2010). More information on ROS events would further improve flood casting and assessments of floods (McCabe, Clark, and Hay 2007).

HUMAN IMPACTS ON FLOODS

Humans have greatly modified the storage capacity of rivers to mitigate flood waters through the destruction of wetlands and the severing of flood plains from their channels (Poff 2002). Impacts of floods are likely to change with shifts in climate. Climate change can greatly impact the abiotic structure of rivers and streams. Climate projections suggest that the occurrence of high-intensity rainfall events will increase in many areas in the future (Lin et al. 2020). An increase in high intensity rainfall would affect ambient flow conditions which dictate the changing type, quantity, and quality of the physical habitat in which organisms live (Bunn and Arthington 2002). Climate change will also increase the frequency and intensity of extreme weather events (Kvočka, Falconer, and Bray 2016).

Socio-economic factors which may impact flood potential include changes in population size and urban development. Changes in the socio-economic system may alter the consequences of future floods (Rojas, Feyen, and Watkiss

2013). Construction in floodplains, channel straightening, building of dikes, dams, and construction activity generating impermeable surfaces are examples of human influences that increase the risk of river floods (Daniel, Florax, and Rietveld 2009). An increased likelihood of floods would have a positive correlation with flood return periods. A flood return period is the probability that a flood will occur in any given year.

2019 FLOODING BACKGROUND

In 2019, Nebraska streams and rivers had the worst flooding in decades, and in some locations the worst flooding on record occurred (US Department of Commerce, 2020). Every month in 2019 except February, had an area based monthly runoff in the 95th percentile (USGS, 2020). In winter storm Ulmer, temperatures across the state rose to 15.56 degrees Celsius with 0.06 mm of rain in a three-day period (March 11 - March 13) (Omaha, NE Monthly Weather). Nebraska experienced a 34 millibar pressure drop within a 24-hour period leading to the creation of a bomb cyclone, where the average center pressure falls at least 1 mb h^{-1} for 24 hours (Frederick and Gyakum 1980). With increasing flows due to excess runoff, ice in the rivers created ice jams and large slabs of ice were deposited and forced onto bridges and roadways (US Department of Commerce). Some ice chunks were estimated to weigh around three tons (Nebraska Flooding, 2019), and contributed to the collapse of the Spencer Dam on the Niobrara River, destroying three bridges downstream of the dam.

Downstream of the Niobrara, residents along the Missouri, Platte and Elkhorn Rivers were forced to evacuate The Missouri River flood exceeded 14 meters in some areas, displacing around 4,400 people in the region (Resnick, 2019). Over 300 kilometers of roads were damaged and 16 state highway bridges were unpassable (Spiegel, 2019). Forty-one breaches to federal and non-federal levees were reported by the U.S. Army Corps of Engineers (Nebraska Flooding, 2019). Nebraska suffered $1.3 billion in property damages, including $449 million in damage to roads, levees, and other infrastructure, $440 million in crop losses, and $400 million in cattle from the 2019 flood (Schwarts, 2019). Three people from Nebraska and Iowa died from this historic flood (Schwarts, 2019).

It takes years to rebuild infrastructure damaged by floods (Croope and McNeil 2011) and some farms and business may never recover. Destruction and damage to businesses can result in the loss of jobs, negatively affecting incomes and thus creating even greater challenges to recover from disasters (Tierney 2007). The social repercussions from this flood are still undetermined, and even less is known about the biotic and abiotic outcomes.

Nebraska presents an optimal case study to examine how flooding resources can be utilized to enhance our understanding of floods caused by extreme weather events. Understanding the vulnerability of societies, both ecological and human, is crucial for understanding historical trends in flood risk and producing accurate projections of fatalities and losses (Brenden Jongman et al. 2015). Currently there are only a few resources available to quantifying the extent of floods. Furthering our knowledge can help mitigate deleterious consequences for people that live near rivers that are destined to encounter. For example, after the March 2019 flood in Nebraska, one farmer had a deposition of 3 to 4 feet of sand in his corn field (Ducey, 2019).

Very little is known about the relative importance of rain-on-snow events for the flood hydrology of the western United States (McCabe, Clark, and Hay 2007) and even less is known about the Midwest. This research begins to fill the gap by first designating the March 2019 flood as weather whiplash event and by showing the limited ability to prepare for future events in Nebraska. The objectives of this projective were to 1) quantify the extent of the March 2019 Nebraska flood, and 2) evaluate the factors that caused the extreme flooding.

FIELD SITE DESCRIPTION

Nebraska has a change of latitude 40° to 43° N and longitude 95° 19′ to 104° 3′ W, with the total area of the state being approximately 200,356 km2 (V. Sharma and S. Irmak 2012). Nebraska has 19 ecoregions throughout the state and has over 130,000 kilometers of streams, for which approximately 28,000 kilometers flow continuously (Stream Biological Monitoring Program, 2019). Within the state there are four major river basins- the Niobrara, Platte, Missouri and Republican. Many streams and rivers in the state have been modified to succumb to the large percent of land use of agriculture and need for irrigation. It is the 3rd leading producer nationwide for corn exports and 4th for soybean (Nebraska Department of Agriculture 2019).

The climate of Nebraska can be divided into two regions: the eastern and central parts of the state are humid/sub humid continental climate, and the western third has a semiarid/arid climate (Sharma and Irmak 2012). This creates wide variability in the state’s climate. Nebraska has a large precipitation gradient to where precipitation in the northwest region of the state averages about 360 mm annually and increases to more than 860 mm in the southeast region (Bennett et al. 2007).

Streams in Nebraska have been channeled and straightened impacting their once natural flow and meandering regime. The changing climate, land use and soil all impact Nebraska’s ability to mitigate floods. Over the past 100 years, the Upper Midwest, of the United States, has suffered from floods and Nebraska is no exception (Villarini et al. 2011).

FLOOD METRIC

To describe flood conditions, several metrics exist. Metrics, such as return intervals or flood stage, rely on having high frequency and long-term data on stream discharge. Alternatively, some metrics like flooding extent can rely on having unobstructed aerial satellite imagery, which is not always possible due to cloud cover and satellite rotation.

One of the most common methods to describe a flood, the “return interval”, also known as return periods, is based on the probability that a given event will be equaled or exceeded in any given year (USGS, 2020). If a flood has a statistical probability of occurring 0.01 times in any given year it is called a “100- year flood” (USGS, 2020). One-hundred year floods can happen twice in year due to various factors influencing rainfall and streamflow. But, the "100-year flood" of yesterday may not be the same "100-year flood" of today due to land use and climate changes. Return intervals are important in hydraulic design, flood frequency estimations and flood insurance (Saksena, 2017).

Return intervals are calculated using frequency analysis to help quantify changes in ambient stream flow. Frequency analysis uses statistical techniques to estimate the probability of the occurrence of a given hydrological event (Saksena, 2017). The flood frequency analysis relates flood discharge values to return periods to provide an estimate of the intensity of a flood event (Saksena, 2017). A return period is an average time, or an estimated average time between flooding events. Thus, using a plotted flood frequency curve, estimations of repetition can be calculated with the corresponding reoccurrence intervals with a trend line.

Flood conditions can also be described by assessing changes in the flow pattern (Blaszczak et al. 2019). The severity of flooding at a given stage may not be the same at all locations along a stream. This is due to the varying channel/bank characteristics on portions of streams, making flooding a local phenomenon. The stage for a given flood category is usually associated with the lowest water level corresponding to the most significant flood impacts on the reach (US Department of Commerce, & NOAA 2016). The National Weather Service uses three terms used to quantify high water: bankfull stage, action stage, and flood stage. Bank full stage, associated with the 2-year return interval, is an established gage height at a given location along a river or stream, above which a rise in water surface will cause the river or stream to overflow the lowest natural stream bank somewhere in the corresponding reach (US Department of Commerce, 2016). A flood stage is an established gage height that creates a hazard to lives or property (US Department of Commerce, 2016). Within the flood stages there are three categories defined for each gage location that describe the observed/expected severity of flood impacts within the corresponding stream segments. A minor flood has minimal or no property damage, but possibly some public threat. A moderate flood has some inundation of structures and roads near the stream and some evacuations of people and/or transfer of property to higher elevations may be necessary. A major flood has extensive inundation of structures and roads where evacuations of people and/or transfer of property to higher elevations are necessary.

Caution should be used when interpreting hydrograph data to due to the impacts of flooding. Missing values, spatial variance and short term data sets can alter the results (Kundzewicz, Hirabayashi, and Kanae 2010). A big enough flooding event could wash out the monitoring equipment thus enabling the discharge values to be adequately recorded- so this event would not calculate accurately.

NEBRASKA FLOODING

To quantify the impacts of the March 2019 floods in Nebraska, I used two forms of hydrological climate data: return intervals- and pixilation. To more closely examine potential causes of flooding related to the ROS event, I also evaluated climate data in two watersheds within the state that had flooded USGS stream gages.

HYDROLOGICAL DATA ANALYSIS

Return intervals were calculated for each USGS stream gage station using hydrological data collected and following the methods of flood analysis (Klingeman, Peter). The return interval analysis uses a log-Pearson Type III calculation to determine the frequency for each stream monitoring locations using the following equation (U.S. Water Resources Council 1982):

$$\log(Q_{Tr}) = avg(\log(Q) + [K(Tr \times Cs)] \times \sigma logQ \qquad \text{[Equation. 1]}$$

where Q_{Tr} is the discharge for the return period (m^3/s), Q is the peak flow for each year (m^3/s), Tr is the return period (yr), and K is the frequency factor, which is a function of the skew coefficient, Cs, and the frequency factor table. Cs is calculated by:

$$Cs = \frac{n \times \sum_i^n (\log Q - (avg(logQ))^3}{(n-1)(n-2)(\sigma logQ)^3} \qquad \text{[Equation 2]}$$

where

$$\sigma \log Q = \sqrt{variance} \qquad \text{[Equation 3]}$$

and

$$variance = \frac{\sum_i^n (logQ - avg(logQ))^2}{n-1} \qquad \text{[Equation 4]}$$

Out of 134 USGS stream gage station sites, 40 were missing discharge data during March 2019 and were omitted from this analysis. DNR stream gage stations were not include in this analysis. Results from return intervals included 94 USGS stream gage stations, that were then analyzed in ArcGIS (ver. 10, ESRI, Redlands, Cal.) using spatial interpolations such as IDW and quantity graduated symbols to better visualize differences throughout the state.

To measure the spatial correlation, I used the Global Moran's I test in ArcMap. This metric helps quantify how close return interval values are to each other. Global Moran's I test measures spatial autocorrelation based on feature locations and attribute values simultaneously. A positive Moran's I index value, which occurs when the z-score or p-value has statistical significance, indicates tendency toward clustering while a negative Moran's I index value indicates tendency toward dispersion (Balaji and Saravanabavan 2020).

Through hot spots identification, natural resources managers can focus limited resources in areas where flooding is most likely to occur. Therefore, I tested the return interval analysis for hotspots using the Optimized Hot Spot Analysis. The Optimized Hot Spot Analysis in ArcMap identifies statistically significant spatial clusters of high values (hot spots) and low values (cold spots). It creates an Output Feature Class with a z-score, p-value, and confidence level bin field (Gi_Bin) for each feature in the Input Feature Class. The z-scores and p-values indicate whether the observed spatial clustering of high or low values is more pronounced than one would expect in a random distribution of those same values. The higher (or lower) the z-score, the more intense the clustering. A z-score near zero indicates no apparent spatial clustering (Du et al. 2017). A feature with a high value is interesting but may not be a statistically significant hot spot.

To be a statistically significant hot spot, a feature will have a high value and be surrounded by other features with high values as well. The local sum for a feature and its neighbors is compared proportionally to the sum of all features; when the local sum is very different from the expected local sum, and when that difference is too large to be the result of random chance, a statistically significant z-score results. For statistically significant positive z-scores, the larger the z-score is, the more intense the clustering of high values (hot spot). For statistically significant negative z-scores, the smaller the z-score is, the more intense the clustering of low values (cold spot) (Mitchell, 2005).

Using the National Weather Service information of flood gage height, flood stage data was compared across streams to determine how many streams reached flood stage.

PIXILATED IMAGE ANALYSIS

To determine the extent of floods, I used pixelated image analysis. Pixelated image analysis helps mitigate many issues associated with complex image analysis. Complex images containing multiple objects and poor quality images can create noise and false attributes (Sowmya and Trinder 2000). The color contrast of bands with snow and ice within the rivers could were extremely difficult to differentiate from the soil. The realization of intelligent recognition of different ground targets are still difficult; no strong image processing algorithm can be used (Chen et al. 2016). Methods of knowledge representation and modelling are still under development making it inconclusive to make firm interpretation automation of information extraction from aerial and satellite images (Sowmya and Trinder 2000). With better spatial resolution and more readily available LIDAR, the uncertainties will decrease (Hostache et al. 2009). As methods are being improved in the extraction of information from digital images for data acquisition, researchers are collaborating with machine learning experts to gain a better understanding (Sowmya and Trinder 2000).

NASA Goddard's Hydrology Laboratory, with combined efforts of the Darmouth Flood Observatory (DFP), created a near real-time global flood map using available satellite data. The pixilated images were sourced from the National Aeronautics and Space Administration near Real Time (NASA NRT) Global Flooding Mapping Basic System. Onboard NASA's Terra and Aqua satellites, the Moderate Resolution Imaging Spectroradiometer (MODIS) instrument provides twice daily near-global coverage at 250 m resolution in two optical bands. This coverage is used to quantity base flow and flood. Using a water detection algorithm developed by Bob Brakenridge of the Dartmouth Flood Observatory, flood levels were detected using optical bands and compared them to ambient water levels. The detected water is compared to a reference water layer that shows "normal" water extent. Pixels found outside the normal water extent are marked as a flood.

March 22nd and 23rd, 2019 was chosen for the pixilated image because it was after the major dam collapses in the state. Although I suspected this date to have the most amount of flood water, I could have missed flood data from flashflood or naturally melted water able to percolate into the ground that occurred prior to this date. The pixilated images, from a 2-day composite, was joined with a Nebraska stream layer (Nebraska Department of Natural Resources (DNR)), to show the interconnections of the streams and then bolded. These pixilated values were later joined in ArcGIS (ver. 10, ESRI, Redlands, Cal.) for with Nebraska HUC 10 shape files (Nebraska DNR), to represent the flooding extent at a watershed scale. This information quantified the severity of the flood across the various regions in the state.

IVESTIGATION INTO POTENTIAL CAUSES OF FLOODING VARIABLILITY

To evaluate the factors that caused the extreme flooding I examined climate and land use data for two watersheds in Nebraska (Figure 1): the upper watershed containing Bazile Creek (hereafter, “Bazile”) and the lower watershed containing Little Blue River (hereafter, “Deweese”).

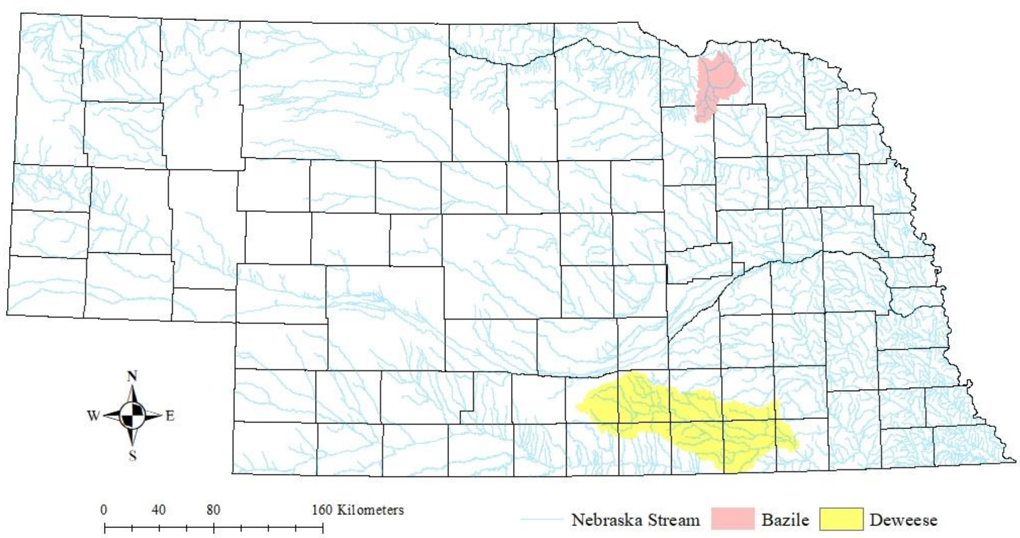

Figure 1 Bazile and Deweese watersheds used for the case study.

Climate data including precipitation, temperature, and snow fall and snow depth was downloaded from the High Plains Regional Climate Center (HPRCC). Soil temperature at 10.1 cm was downloaded from the Automated Weather Data Network (AWDN).

To determine potential influences of the flood soil characteristics for each watershed were collected from Web Soil Survey (NRCS, 2019). Soils were defined by hydrological class as being either class A, soils with a high infiltration rate, class B soils have a moderate infiltration rate when thoroughly wet, class C soils have a slow infiltration rate when thoroughly wet, or class D soils with a very slow infiltration rate when thoroughly wet (USDA NRCS, 2016). When wet, class D soils would have the highest potential for runoff.

Land use was obtained from the National Agricultural Statistics Service using the CropScape- 2019 cropland data layer. Using ArcMap, the 2019 land use and area for each watershed were calculated. Land use was reclassified into six categories: grassland, urban, row crops, forest, non-row crops, or water. The slope of the watershed was calculated using the DEM and using the slope function in ArcMap (NRCS, 2020). Using natural neighbor, I extrapolated the climate data to fit our watershed scales. The natural neighbors of any point are those associated with neighboring Voronoi (Thiessen) polygons (ESRI, 2020). This data provided the visual for spatial change of the climate data gather from the climate stations.

There were five climate stations used to interpolate the climate data in Bazile and one soil sensor used due to missing available data. In the Bazile watershed, the HPRCC climate station Bloomfield (250945 Coop) in Knox County was in between the USGS stream gage stations.

There were 14 climate stations used to interpolate the Deweese climate data and two soil sensors used. In between the two USGS stream gage stations in Deweese was HPRCC climate station Hebron (253735 Coop) in Thayer County. For the climate data I used the average daily values.

RESULTS

Out of the 134 continuous USGS stream gages within the state, return intervals were calculated for 94 streams (Figure 2). The average return interval was 48 years ± 73 years.

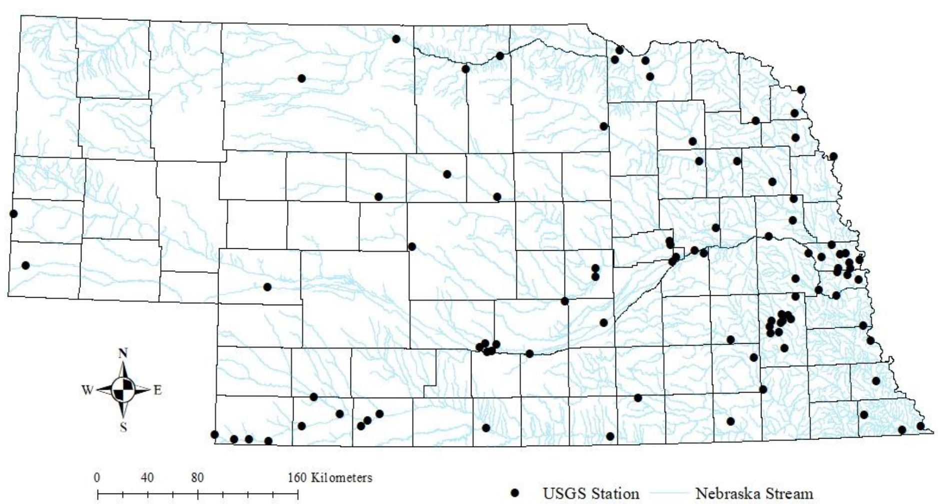

Figure 2 USGS stream gage stations used for the flood return periods.

Of the total amount, 31 sites had the highest flow rate in 2019, and 17 stream gage stations that had over a 100-year flood (Figure 3).

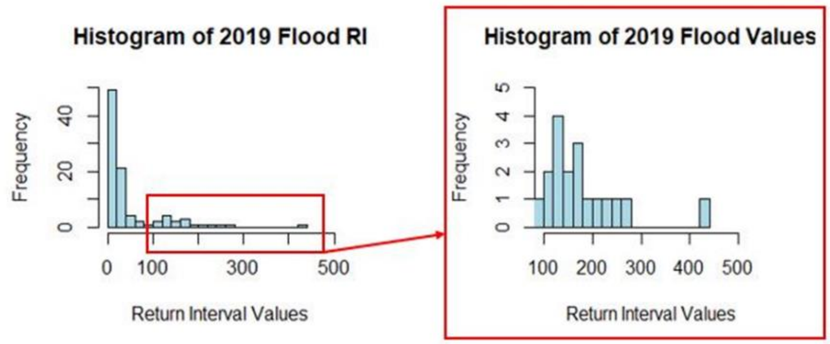

Figure 3 Histogram of 2019 flood return periods.

The return intervals varied significantly within the state. In using the graduated symbol, the highest return values were along the along the Platte, Niobrara and Elkhorn River (Figure 4).

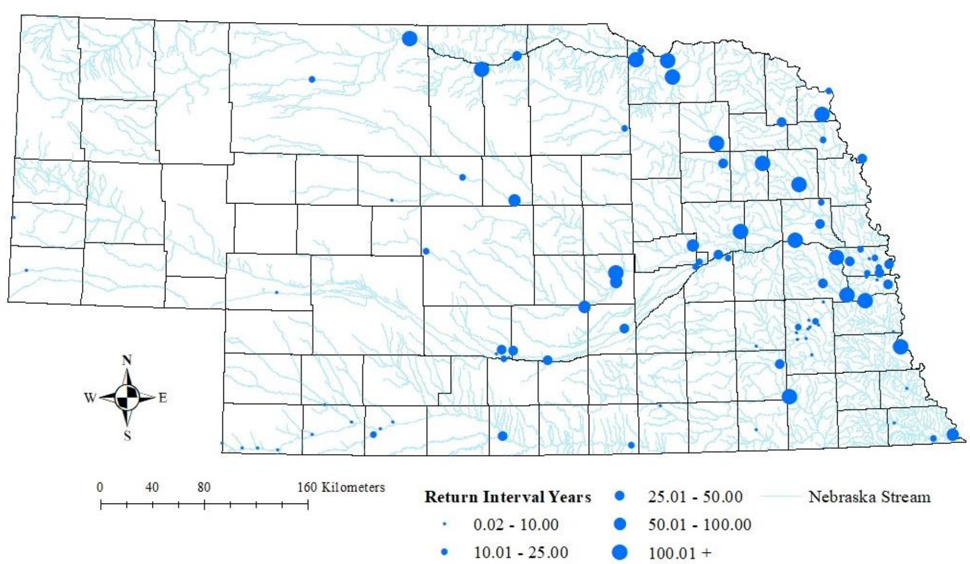

Figure 4 Graduated symbol results from 2019 flood return periods.

The IDW interpolation showed the highest return intervals being in the northeastern part of the state (Figure 5).

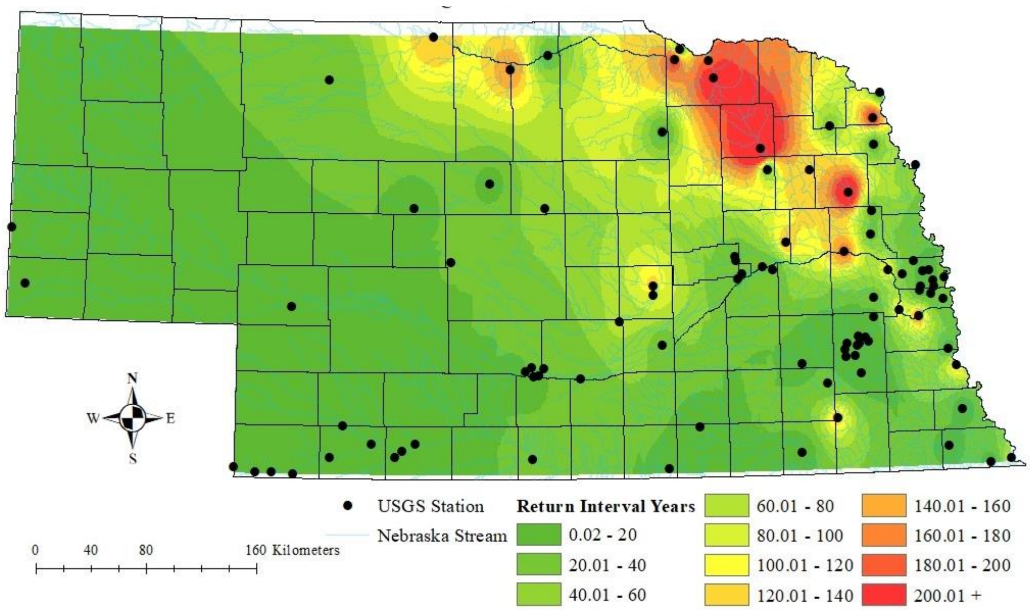

Figure 5 IDW results from 2019 flood return periods.

Eighty-one sites had flood stage data to compare to the March values (Figure 6).

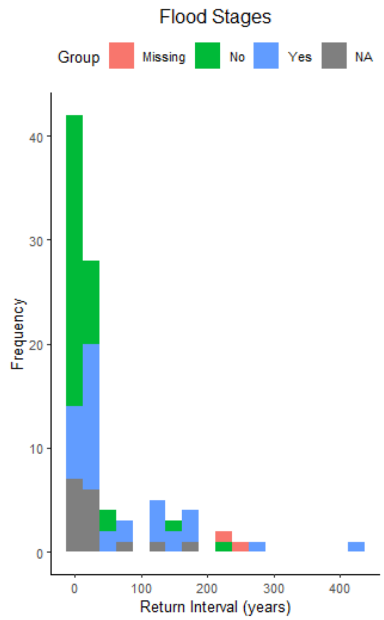

Figure 6 Distribution of flooded gage stations in March 2019.

Forty-six stream gage locations were at a flood stage sometime in March 2019. (Figure 7).

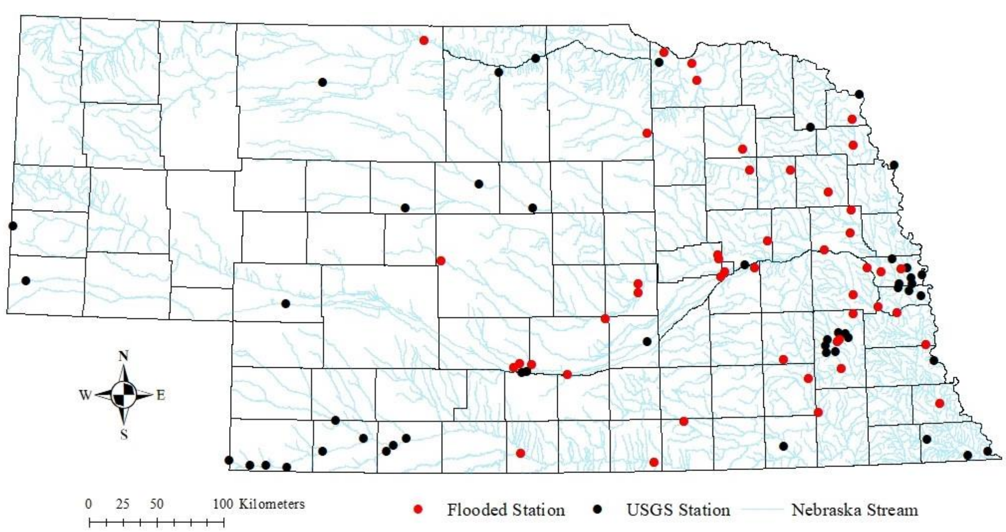

Figure 7 Flooded gage stations in March 2019.

Similar to the return interval analysis, most streams that reached flood stage were along the Platte, Niobrara and Elkhorn Rivers. Results from the Global Moran's I test suggest that the spatial distribution of the return interval vales were not random (Index = 0.15, Z score >3 and p <0.01). Values were clustered and show connectivity to the flooded streams. Streams and rivers, regardless of size or frequency of flow, are connected to downstream waters and strongly influence their function (US, 2015).

Optimized Hotspot analysis resulted in 11 locations found to be significant in the 2019 return interval values. Seven locations had a 95% confidence interval and four had a 90% confidence interval (Figure 8).

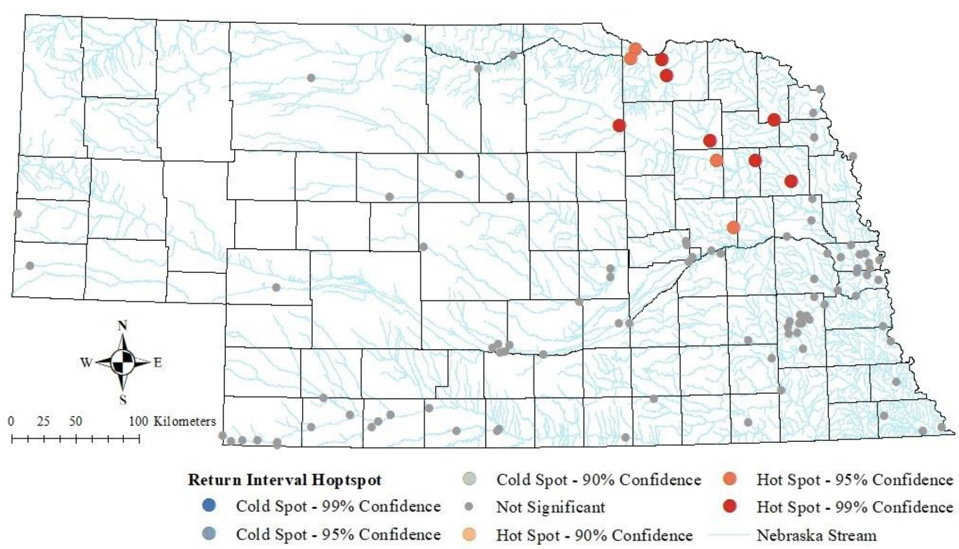

Figure 8 Optimized hotspot analysis from 2019 flood return periods.

These locations had a high z-score ($Z > 3$) and small p-values ($p < 0.00$) indicating a spatial clustering of high values. The hotspots are along the Elkhorn and Niobrara Rivers. Although the spatial distribution and hot spot analysis was conducted using the return intervals of USGS stream gage stations, if I used other gage stations that were missing from our results, this could have skewed or improved. Mapping the reach of rivers and streams that are at bank full stage or over flowing into the floodplain during floods is in high demand (Fujita and Kunita 2011), but our pixilated images were able to show the vulnerability of watershed that had flooded streams. Results from the pixelated image shows that the northeast regions of the state suffered the most from the flood. The Missouri, Platte and Elkhorn Rivers highlighted in Figure 9, had continuous high flood levels (Figure 9).

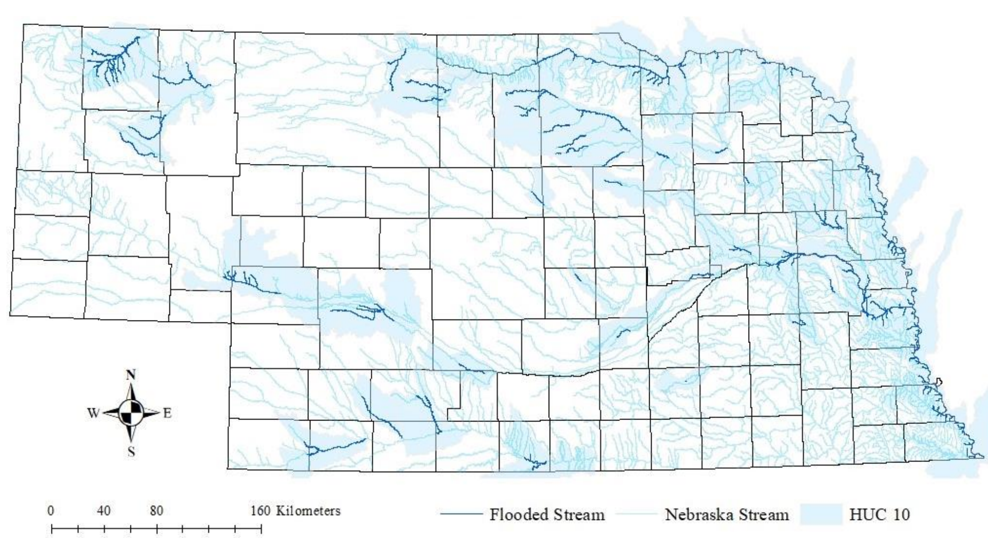

Figure 9 Pixilated flooded streams and watershed from NASA NRT.

TWO CASE STUDIES: BAZILE AND DEWEESE

BAZILE

Bazile Creek has two USGS stream gage stations near Niobrara and at Center. Bazile near Niobrara had a 177 year flood in 2019 while Bazile at Center had a 259 year flood. Both USGS stream gage locations flooded sometime in March.

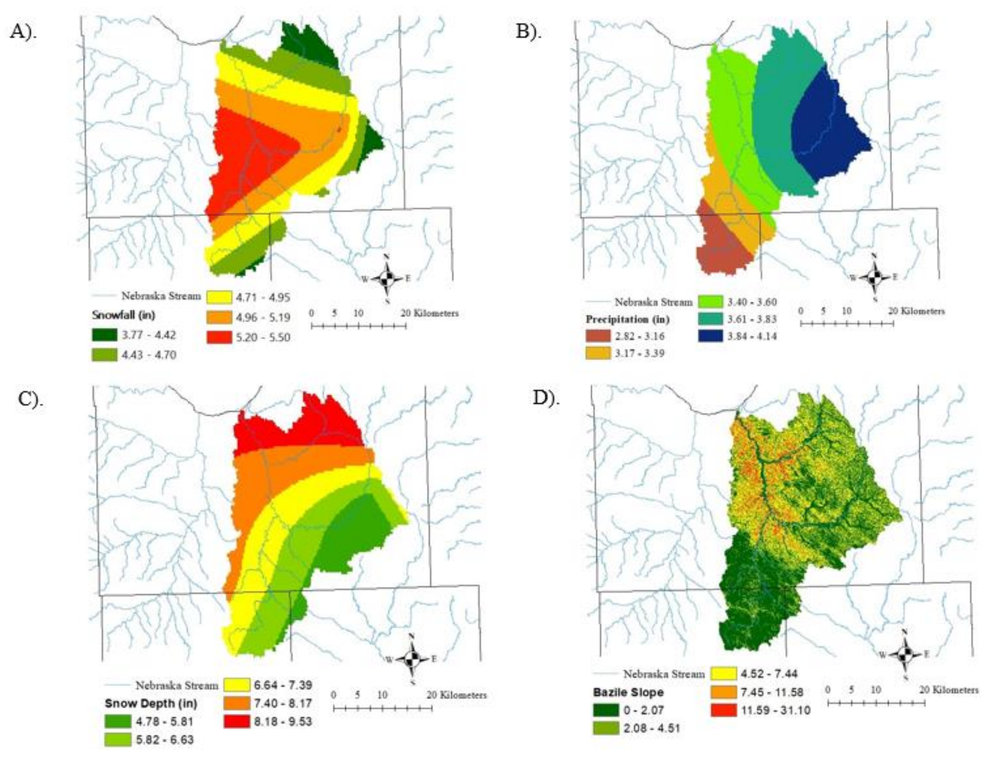

Figure 10 Bazile climate data.

Of the two watersheds, Bazile had a varied soil hydrological class distribution. Bazile watershed has an area of 1152 km^2 with 33% of the soil being a hydrological class C, followed by 30% class B and 26% class A. 47% of the land use is row crops followed by 33% grassland. For March, the Bazile watershed had a maximum soil temperature range of -26.12 °C to 4.82 °C. The maximum air temperature ranged from -24.07 °C to 21.29 °C (Figure 10).

Using Bloomfield's climate data a combined graph was created to analyze peaks in climate data. The highest cumulative precipitation were in the eastern area of the watershed (up to 105 mm), while the highest snow fall was found in the western region (up to 139 mm). The deepest snow was found in the north (up to 242 mm). Topographically, Bazile's steepest region is in the mouth of the watershed. As the snow depth decreased (dark blue), on March 13th the stream gage did not reach a flooded interval (red) simultaneously with the increasing trend of air temperature (orange) and precipitation (light blue) (Figure 11). The brown line is showing the soil temperature at 10cm while the gray horizontal lines are showing the freezing/thawing temperature. Bazile at Center is missing daily gage height data but according to continuous data from the USGS Bazile at Center did flood. Figure 12 shows the same color trend and the gage height did surpass the flooded height.

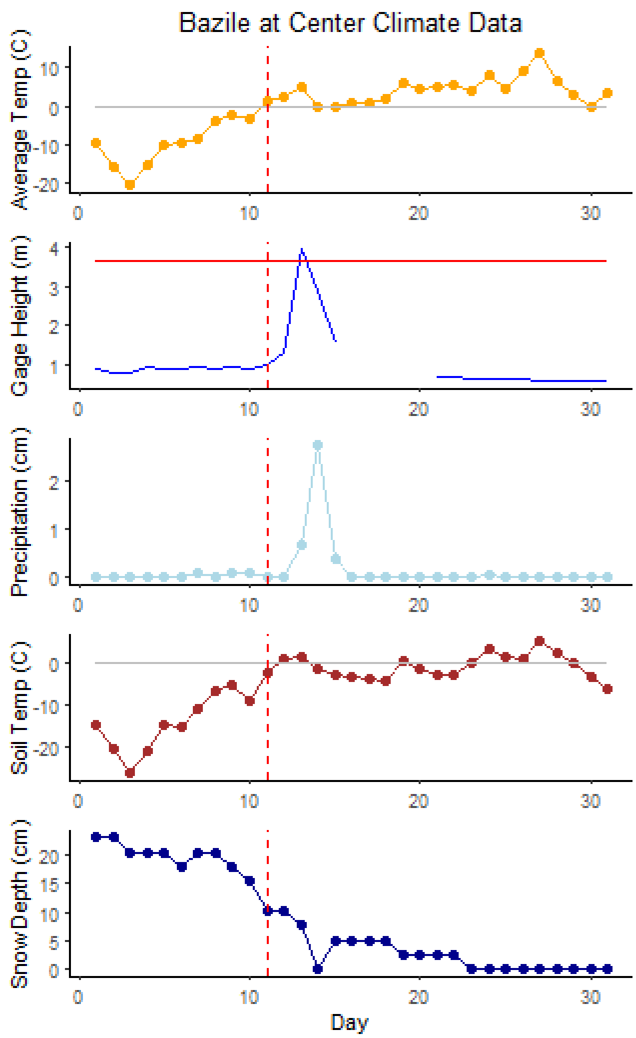

Figure 11 Bazile at Center climate data with USGS stream gage data.

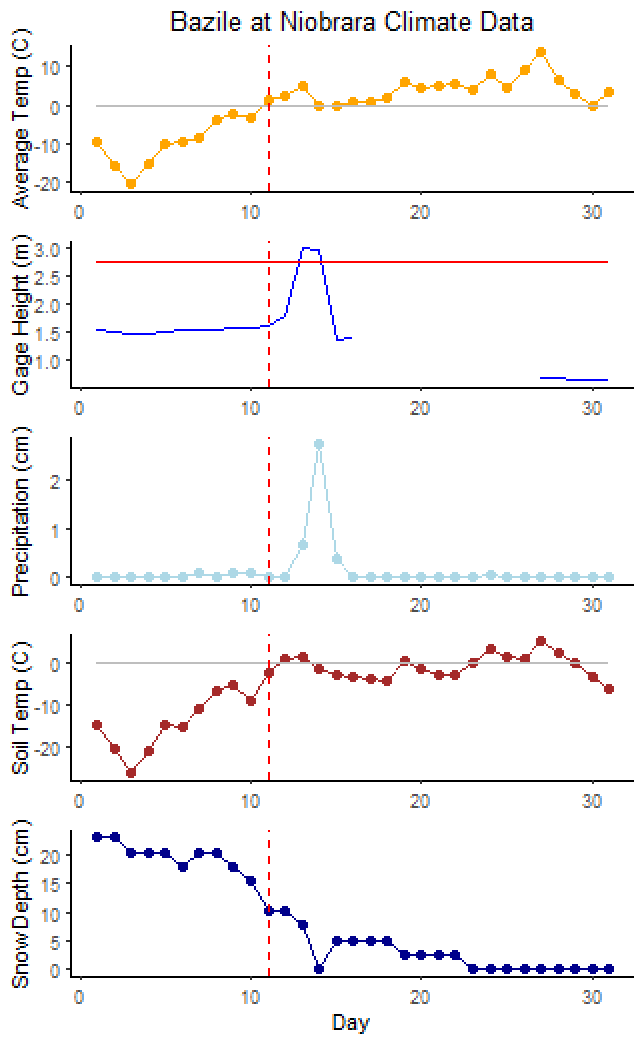

Figure 12 Bazile at Niobrara climate and USGS stream gage data.

DEWEESE

Deweese watershed had Little Blue River near Deweese (USGS ID 6883000) and Little Blue River near Fairbury (USGS ID 6884000) USGS stream

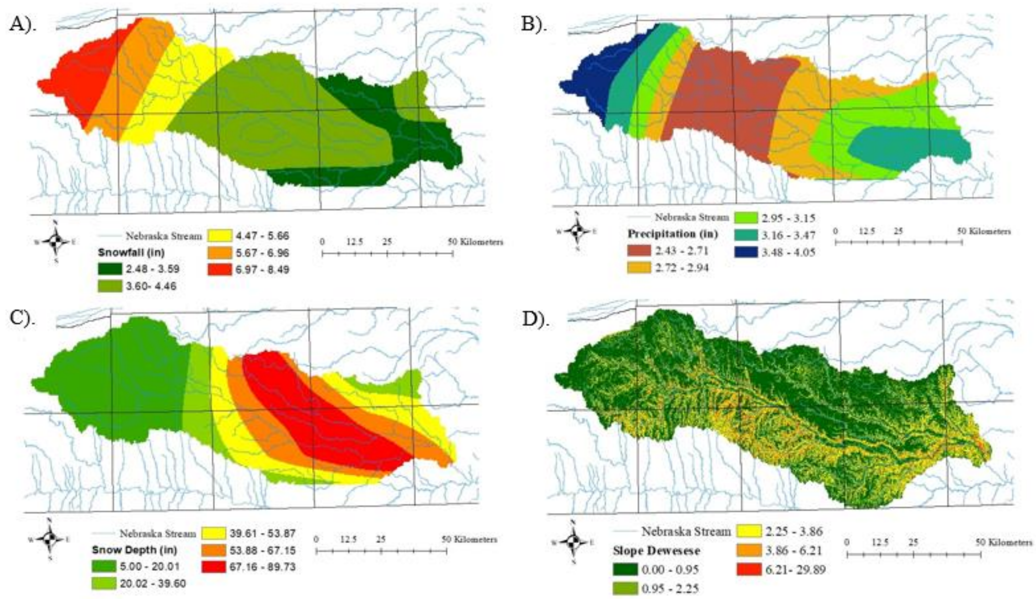

Figure 13 Deweese climate data.

gage stations. Little Blue River near Deweese had a 2 year flood in 2019 while Little Blue River near Fairbury had a 5 year flood. Little Blue River near Deweese was flooded sometime in March.

Deweese watershed has an area of 6,068 km^2 with 50% soil being a hydrological class C. Sixty-nine percent of the land use is row crops followed by 17% grassland. For March Deweese had a soil temperature range of -24.07 °C to 8.78 °C. The air temperature range was -23.89 °C to 24.53 °C. Deweese had the highest slope in the southern regions of the watershed (Figure 13).

The highest cumulative precipitation (up to 102mm) and snow fall (up to 215 mm) occurred in the western area of the watershed. The deepest accumulative snow fall (up to 2,279 mm) was in the eastern region of the watershed. Using Hebron's climate data and Little Blue River near Deweese, USGS ID 6883000, (Figure 14) and Little Blue River near Fairbury, USGS ID 6884000, (Figure 15) a combined graph was created to analyze peaks in climate data. At Deweese, the snow depth was decreasing (dark blue), on March 13th the stream gage reached a flooded interval (red) simultaneously with the increasing trend of air temperature (orange) and precipitation (light blue). The brown line is showing the soil temperature at 10 cm while the gray horizontal lines are showing the freezing/thawing temperature.

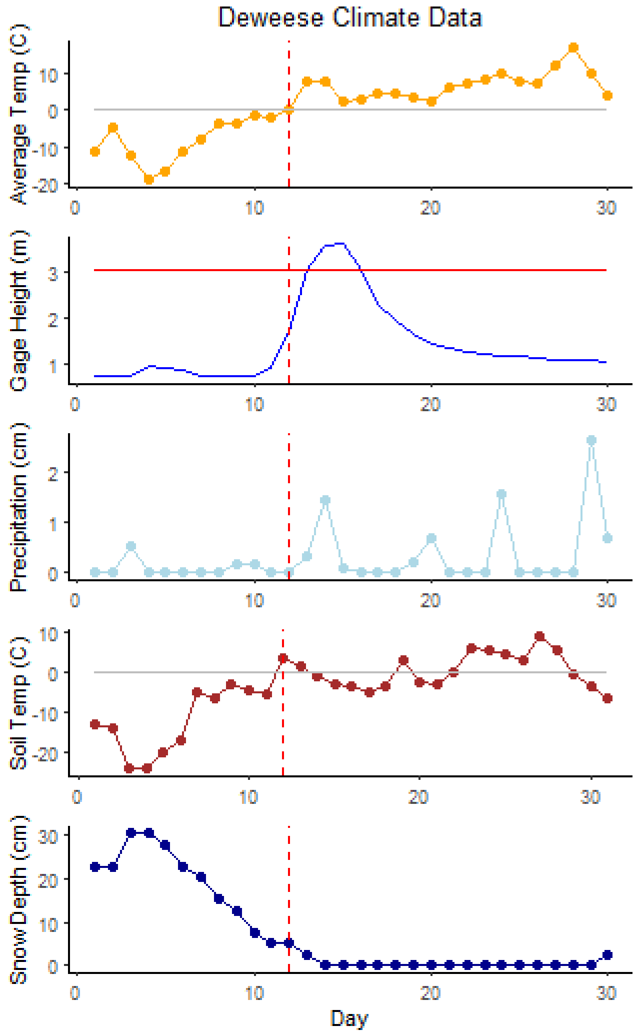

Figure 14 Deweese climate and USGS stream gage data.

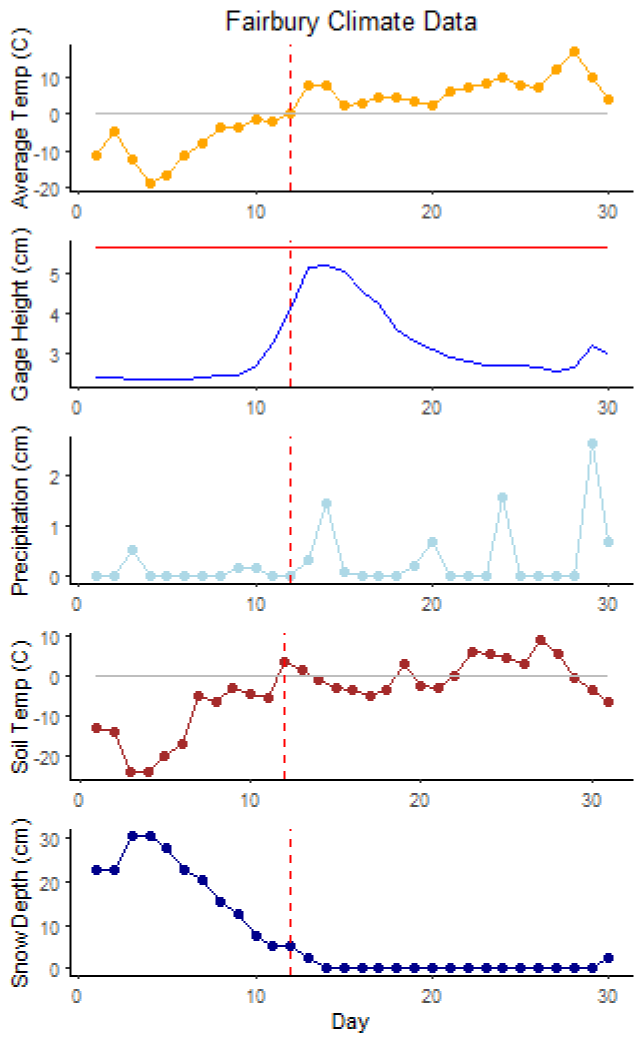

Figure 15 Fairbury climate and USGS stream gage data.

For Fairbury the color schemes are the same for the climate data but the USGS stream gage did not flood. Because Fairbury and Deweese had similar climate features I explored some abiotic features of the Little Blue River, which flows between both sites. The Little Blue River is a sandy meandering stream which naturally slows the flow of water when compared to channelized rivers. In the Deweese watershed there are eighty six dams (Nebraska DNR), and between the two USGS stream gage stations, fifty six dams are present (Figure 16). Within Nebraska there are 2,946 dams (Nebraska, DNR) (Figure 17). Most of the dams are along the southeastern part of the state.

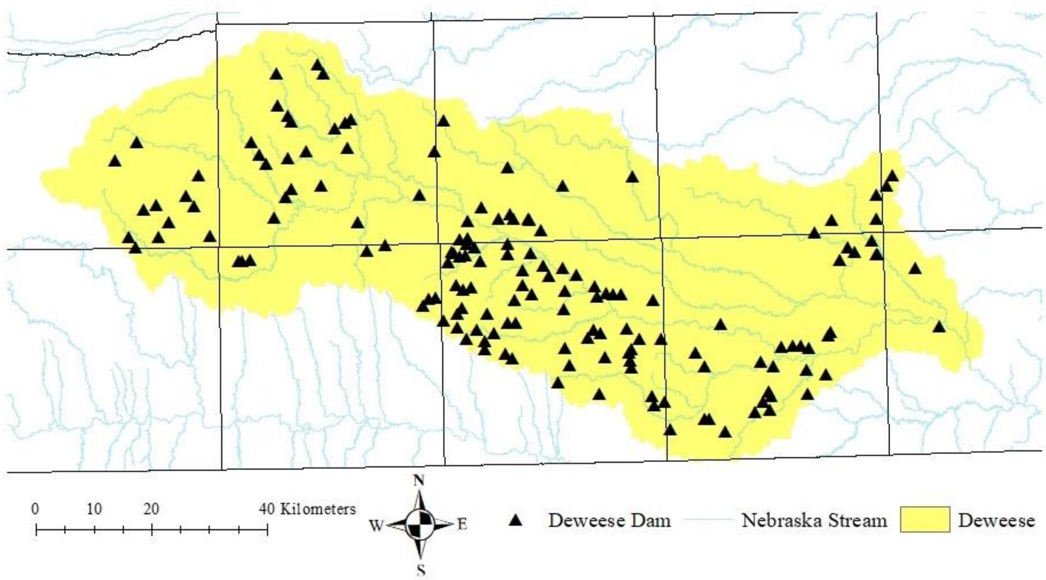

Figure 16 Deweese watershed with dam locations.

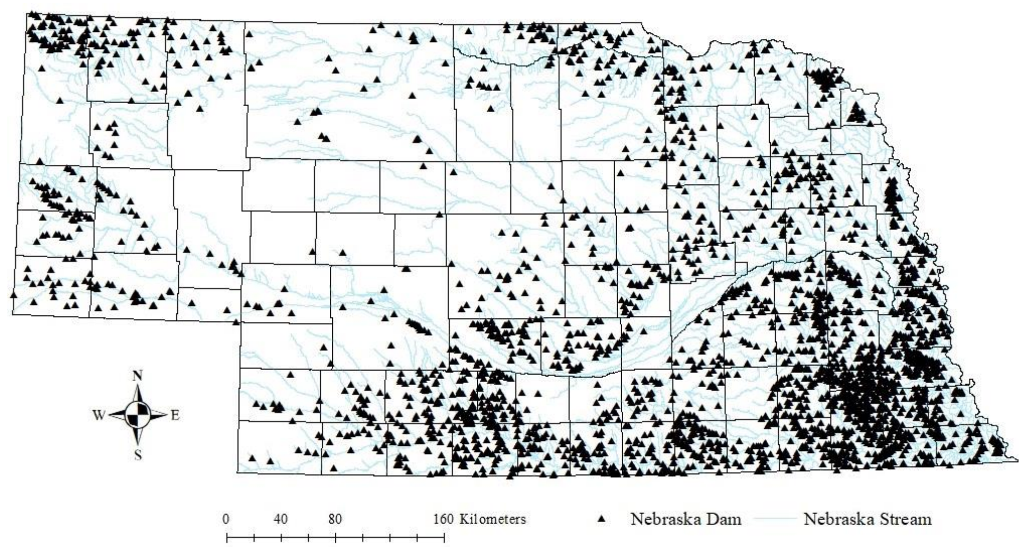

Figure 17 Nebraska dam distribution.

DISCUSSION AND LIMITATIONS

In quantifying the extreme flood, more than half of the Nebraska streams had a flooded gage height. The National Weather Service determined flood gage height for streams across the nation but this brings to light how often are these streams re-evaluated for changes in catchments that affect the impact of floods? More resources should be implemented into reevaluating stream flood height one a more frequent basis. Changes in local land use, river impoundments, fluctuations in total impervious surfaces, and long term patterns can all affect at what point a "100-year flood" is designated (Floods, 2020). The development of an urban area within a catchment is a drastic change of land use that effects the function of the hydrological cycle during flooding conditions (Hollis, 1975). Most gages in remote areas may not have a major flood stage assigned. In remote areas with few specific impacts, floods with 50-100-year return interval would be assumed to be causing major flooding on streams in the area.

There are various technologies that can show the extent of a flood, but each has limitations. Difficulties in quantifying the extent of the flood using aerial images proved to be insufficient to calculate statewide impact in the case of the 2019 Nebraska flood. Cloud and snow cover impaired the water calculations leaving multiple areas of the state obsolete for data analysis.

In calculating return intervals, 14% of sites were missing data due to human and natural forces. Multiple sites only had a limited amount of years with data, which could skews the results. Accuracy of the return intervals varies depending on the amount of data available, the accuracy of data, land-use changes in the river watershed, and climate cycles. Limited locations of data prevented a fully detailed statewide impact on the 2019 flood. The areas that did have useable data did show a huge impact from this flooding event. Locations of where return intervals are located limit the result of a larger analysis of a flood. The return interval analysis also assumes that the probability of the event occurring does not vary over time and is independent of past events.

In our case study to evaluate the factors that caused the extreme flooding, both watersheds had a rain on snow (ROS) weather event, which I deem to fit a winter weather whiplash event. The precipitation on top of snow and frozen ground had a synergetic effect of enhancing the flood. Changes in temperature, not changes in total annual precipitation affect the frequency of ROS events (Casson, Eimers, and Buttle 2010). In our study I did not examine the damaged infrastructure or the ice chunks in the rivers, which could have also influenced the flood. I estimate that the presence of dams within Deweese watershed could have slowed the flooded watershed protecting Fairbury USGS stream gage station. The amount of physical infrastructure in watersheds could influence the flood susceptibility of locations. Deweese watershed had more dams and a lower return period for both USGS stations. With the Bazile watershed only having four dams, it was more susceptible to a higher flood return period.

Because of the varied climate zones in Nebraska, I experience drastic temperature and climate changes, enabling ROS events to become more prevalent. A ROS event has never been documented in Nebraska before 2019 thus this is an interesting scientific observation for which I have limited resources to understand the full effects of this event for years to come.

CONCLUSION

When I quantified the extent of the flood I found that not every stream was impacted equally, but many locations on the Platte, Elkhorn, Niobrara, and Missouri River suffered. Extreme events, such as floods, should not be considered stationary. In many locations the hydrologic records exhibit nonstationary trends in data in the form of continuous trends or abrupt shifts (Obeysekera and Salas 2014).

When I evaluated the factors that cause the extreme flooding, I found that a ROS event lead to the quick, extreme flood in Nebraska. Additional research into rain-on-snow events should improve both flood forecasts and assessments of flood risk (McCabe, Clark, and Hay 2007). The projected changes in flood frequencies illustrate the importance of re-evaluating the adequacy of water resources infrastructure and design. This is especially true under conditions where high frequency events become more frequent (Surfleet and Tullos 2013). Knowledge of how much river basins are receiving rainfall and contributing runoff in near-real time would provide a better understanding for forecasts (Kattelmann 1997).

The causes of flooding and their consequences can vary, giving precedence need for increased, internationally coordinated efforts to enhance technologies and policies for flood preparation and response (Djordjevic, Butler, and Gourbesville 2013). Making decisions on flood management should involve expert water managers and other specialists, such as urban planners, architects, engineers, and geologists. Solutions to flood protection do have broad conclusions; every case should be treated separately to ensure proper decision making. Earlier warning systems, increased collaborations with state and federal organizations, and better communication between communities would help detour future negative implications from natural disasters. Furthering our knowledge of flooding impacts on ecosystem process can help mitigate deleterious consequences by enabling an improved warning system for future floods.

CHAPTER 3: EXAMINING THE IMPACTS OF THE 2019 NEBRASKA FLOOD USING NUTRIENT SPIRALING

ABSTRACT

In March 2019, a bomb cyclone in the Midwest United States led to extreme flooding in the region. Nebraska was hit particularly hard, with many streams and rivers overflowing their banks and remaining elevated for months after the event. Here, I compare nutrient spiraling metrics in summer in ten streams in 2018, before the flood, versus 2019, after the flood. Our sampled streams ranged across ecoregions and land uses. I found that there was no significant difference in spiraling metrics such as nutrient uptake, uptake velocity and aerial uptake, pre-and post-flood. Some surprising results show that although nutrient spiraling metrics did not change, stream water concentrations of ammonium (NH_4), soluble reactive phosphorus (SRP), and dissolved organic carbon (DOC) did change. While NH_4 and SRP concentrations decreased, DOC increased after the flood. These data suggest Nebraska streams were somewhat resilient to the historic flooding impacts, but that biologically available nitrogen (N) and phosphorus (P) decreased. As climate change is expected to lead to more frequent large-scale and intense hydrological events, studies like this will help understand how these events may impact aquatic ecosystems and their in-stream processes.

INTRODUCTION

Streams and rivers have the ability to reproduce and/or transform nutrients (Palmer and Ruhi 2019). Nutrients, like nitrogen (N) and phosphorus (P) can be transported from terrestrial and upstream sources through drainage networks and move downstream (Aguilera, Marcé, and Sabater 2013).). Streams and rivers have the ability to become saturated with nutrients, exporting excess nutrients downstream (Small, Helton, and Kazanci 2009). Indeed, while N and P are critical to sustaining life, in excess amounts, they can be detrimental to the environment and cause eutrophication. Eutrophication, which is caused by excess nutrients in water ways leading to anoxic conditions, and death to plants and animals, along with other ecological consequences, which is widely studied due to the negative environmental, health, and economic impacts (Chislock, 2013 In the U.S. alone the estimated cost of damage by eutrophication is approximately $2.2 billion annually (Dodds et al. 2009). Understanding how streams cycle nutrients is important to mitigating eutrophication.

The rates by which nutrients cycle in streams are controlled by several processes, both biotic and abiotic. These processes can occur in multiple complex ways (Glibert 2012). Both autotrophs and heterotrophs influence nutrient concentrations in streams, through processes of primary production, respiration, decomposition, egestion and/or excretion (Welti et al. 2017; Cross et al. 2005; Small, Helton, and Kazanci 2009; Grimm and Fisher 1986; Newbold 1992; Ashkenas et al. 2004). While the aforementioned processes tend to impact both N and P cycling, abiotic processes like sediment adsorption tend to impact only P (Hall et al. 2013).

Abiotic conditions of the stream environment such as hydrology, temperature, light, and geomorphology are indirectly involved in stream nutrient retention. A morphological feature that impacts nutrient cycling is bed sediment. By providing adsorption sites, stream bed sediments can act as a sink to nutrients, rendering them unavailable for biological uptake (Grimm et al. 2003; Corman, Moody, and Elser 2016). The impact of stream sediments is also important in highly modified streams (Reisinger et al. 2019). Along with channel geomorphology, residence time has an indirect effect on increasing nutrient uptake from the water column by dictating the exposure to bioactive surfaces associated with benthic substrates (Ensign and Doyle 2006).

Human activities have drastically altered the ambient concentration of nutrients in streams. Phosphate and nitrate are among the most pervasive sources of freshwater pollution globally (McGoff et al. 2017). Specific external stream nutrient sources can include: inputs of contaminated ground water, non-point, and point sources of pollution (National Research Council 2000). Increased channelization and damming of rivers impacts sediment transport and alters biogeochemical cycles that increase diversity at local and global scales (Sabater 2008).

NUTRIENT SPIRALING

Nutrient spiraling is commonly used to investigate nutrient production, transformation, and uptake. It offers a glimpse into stream biogeochemical processes while providing an approach to analyzing structural and functional adaptations in lotic ecosystems (Newbold et al. 1982). Nutrient uptake by autotrophic and heterotrophic processes is commonly assessed with low-concentration additions of nutrients (Roberts, Mulholland, and Hill 2008).

Normally nutrients that are most limited by autotrophic or heterotrophic mechanisms are used in nutrient spiraling, such as phosphate and inorganic forms of nitrogen (ammonium or nitrate), along with a conservative tracer (Hauer and Lamberti 2007). By focusing on N and P, nutrient uptake can be quantified moving downstream. There are three main metrics used to quantify in situ stream nutrient cycling, uptake length (S_w), vertical velocity (V_f), and aerial uptake rate (U).

Nutrient uptake length combines two basic properties of nutrient dynamics of a stream: the rate of nutrient cycling and the degree of nutrient retentiveness (Newbold et al., 1983). Nutrient uptake and remineralization often control dissolved inorganic nutrient concentrations in aquatic ecosystems (Dodds and Whiles 2010; Dodds et al. 2010).

Uptake lengths can determine stream reach demand for dissolved nutrients and are controlled by the interactions of the downstream nutrient flux, and the contact between the water column and the active benthic communities (Davis and Minshall 1999). This means uptake length is largely dependent on the velocity of the water. Slow discharge combined with rapid nutrient uptake produce short nutrient uptake lengths (Weathers et al. 2013).

Although nutrient spiraling has been performed on many small order streams, understanding impacts of extreme flooding on river ecosystem nutrient dynamics is not documented.

FLOODING

Spatial and temporal patterns of many ecological processes in rivers including metabolism are regulated by floods (Bernhardt et al. 2018). For instance high flows can increase suspended sediments, impeding organismal feeding and reproduction (Palmer and Ruhi 2019).

However, seasonal fluctuations in discharge, including moderate flooding, are crucial to many fluvial species (Welcomme 1985), and are important for the establishment and maintenance of diverse vegetation communities across riverine landscapes (Munes et al. 2015). Some of the benefits of floods include groundwater recharge, increased fish production, wildlife habitat, and wetland recharge (Talbot et al. 2018; Galat et al. 1998). Depending in the magnitude, depth and velocity, floods can either be a scouring event, in which sediments and nutrients are removed from a location, or a deposition event, in which sediment and nutrients are deposited.

Stream and river ecosystems are subjected to multiple stressors that can affect lotic structure and function (von Schiller et al. 2017). At local and global scales land use and climate change alter disturbance regimes of rivers and streams (Bernhardt et al. 2018). Anthropogenic climate change increases the risk of natural disasters such as floods, fires, droughts, and prolonged high temperatures (Hyeck-Williams 2019). Climate change has shown to increase the frequency and intensity of extreme weather events, such as flooding (Kvočka, Falconer, and Bray 2016). Due to the increased likelihood that floods will increase in the future we need to how flooding effects the streams and rivers ability to cycle nutrients.

Because flooding is the most frequently occurring natural disaster (Jonkman 2005), a better understanding of how flooding can impact ecosystem processes, can help mitigate potential negative outcomes.

FLOODING IN NEBRASKA

While studies of flooding impacts have revealed novel insights into river ecological processes, studies of extreme flooding impacts are, by nature, less prevalent. Fortuitously, an extreme flooding event occurred in the US Great Plains in 2019, the winter after I had assessed nutrient cycling rates in streams across the state of Nebraska. Therefore, I was able to return to the ten sites and perform the same nutrient cycling rate assays to determine if flooding impacted nutrient cycling in these stream ecosystems.

Multiple locations along the Platte, Elkhorn, Niobrara, and Missouri Rivers were inundated by the 2019 flood. Many of Nebraska streams reached historic record high levels. Over 12% of Nebraska streams had return intervals of over a 100-year flood and 35 streams reached a flood stage in March alone.

OBJECTIVES

In this study I 1) Examined the nutrient uptake metrics between the pre- and post-flooding sampling campaign, and 2) Compared flooding impacts between sampling locations. As flooding can lead to either scouring or depositing of nutrients, I also determined if the flood increased or decreased rates of in stream ecosystem processes. Nutrient spiraling data were collected seven months prior and seven months post flood, providing a unique opportunity to compare pre- and post-nutrient dynamic changes within these streams

MATERIALS AND METHODS

FIELD SITES

To examine nutrient uptake metrics, I used steady state injections of N and P in ten stream reaches in 2018 and 2019 across Nebraska (Figure 18). I used the 2018 nutrient spiraling metrics for pre-flood classification and 2019 as the post-flood ecosystem response.

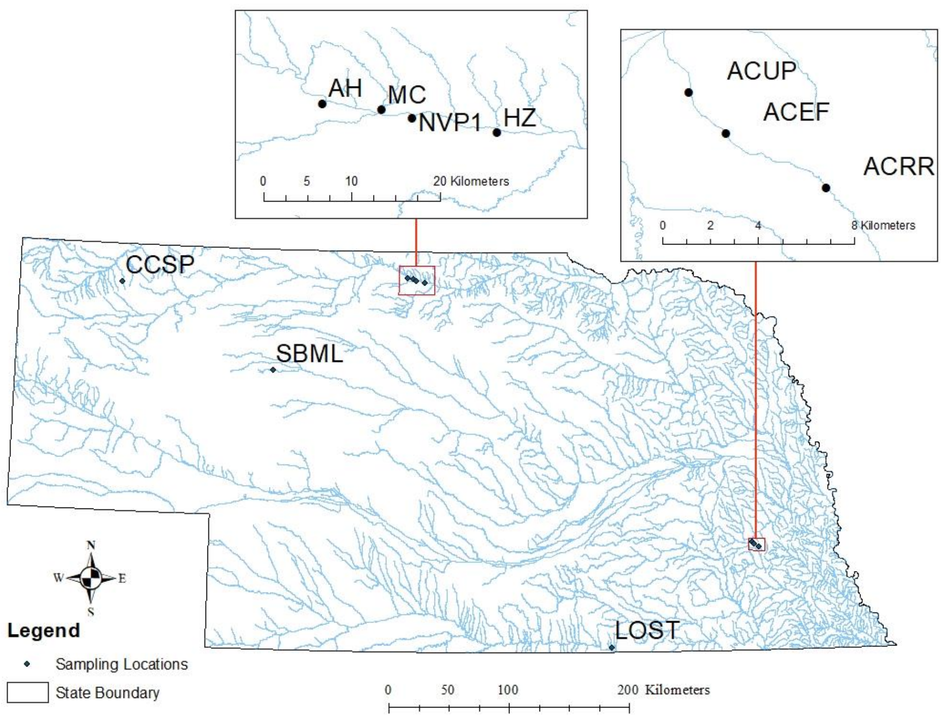

Figure 18 Spiraling locations across Nebraska

I sampled ten streams reaches across Nebraska (Figure 18). Three sampling locations were in Lincoln, NE, the second largest city in Nebraska, Antelope Creek Rickman's Run (ACRR), Antelope Creek Union Plaza (ACUP), and Antelope Creek Elks Field (ACEF). ACRR is located downstream from an urban reservoir called Holmes Lake, and is in a dog park called Rickman's Run. ACUP is located east of downtown Lincoln and is a channelized stream that feeds into Holmes Lake. ACEF is also a channelized stream between a golf course and baseball fields that also drains into Holmes Lake.

In the southernmost part of Nebraska, a sampling location called Lost (LOST), is within a small rural town called Superior, NE. LOST is a within an agriculture crop field that had notable stream embankments.

South Branch Middle Loup (SBML) is located within the Sandhill's of Nebraska within a grazed grassland. The Nebraska Sandhill's are one of the largest plant-anchored sand dune regions in the world, and the largest sand dune formation in the Western Hemisphere. The region is abundant with groundwater fed streams, prairie potholes and range fields.

Chadron Creek State Park (CCSP) is in the Nebraska National Forest in the Nebraska panhandle. The Nebraska National Forest is a 445 thousand km^2 forest conservation & recreation area, plus a historic federal tree nursery. CCSP is within a greenspace in a tourist campground.

Within the Niobrara River Valley of Nebraska there were four sampling sites (NVP1, NVP-AH, NVP-MC, and NVP-HZ). NVP1 is a small tributary stream that flows alongside a residential home. NVP-AH is located within an incised valley within a bison field. MC is located off a gravel road in a floodplain leading to the Niobrara. HZ is also a small tributary stream located in a pasture where bison feed. Streams chosen for nutrient spiraling were selected based on ease of access and to encompass the various land use types throughout Nebraska.

All three Antelope Creek streams are located within the Salt River watershed and the Western Corn Belt Plains ecoregion. LOST is located in the Middle Republican watershed and Central Great Plains ecoregion. SBML is in the Upper Middle Loup watershed and in the Nebraska Sandhill's. All four of the Niobrara streams are in the Middle Niobrara watershed and North Western Great Plains ecoregion, while CCSP is in the Upper White watershed and in the Western High Plains ecoregion (Table 1).

Table 1 Sampling locations with watershed area and watershed names.

Site Name	Watershed Name	Latitude	Longitude	Area (ha)
SBML	Upper Middle Loup	42.0905	-101.45589	$8.84*10^{8}$
NVP1	Middle Niobrara	42.0905	-101.45589	$7.6*10^{5}$
NVP-HZ	Middle Niobrara	42.7675	-99.939084	$2.5*10^{7}$
NVP-MC	Middle Niobrara	42.7907	-100.056588	$4.2*10^{7}$
NVP-AH	Middle Niobrara	42.7968	-100.116522	$1.4*10^{6}$
LOST	Middle Republican	40.038	-98.078721	$4.3*10^{7}$
CCSP	Upper White	42.7099	-103.009181	$3.7* 10^{7}$
ACRR	Salt	40.7735	-96.62225	$2.93 * 10^{7}$
ACEF	Salt	40.7955	-96.670658	$2.93 * 10^{7}$
ACUP	Salt	40.8116	-96.688145	$2.93 * 10^{7}$

WATERSHED AND FLOOD CLASSIFICATION

To determine the magnitude of the flooding at each stream site, I calculated the “return interval”. Return intervals are based on the probability that a given event will be equaled or exceeded in any given year (USGS, 2020). Return intervals are calculated using frequency analysis to quantify changes in ambient river flows. Frequency analysis uses statistical techniques that fit frequency distribution data to predict flooding for a river at some given location (Saksena, 2017).

To compare the impacts of the March 2019 flood in Nebraska, return intervals were calculated for each stream sampling location using the Oregon State University protocol for flood analysis (Klingeman, Peter) and data from adjacent USGS stream gage monitoring stations. The log-Pearson Type III analysis described by the Oregon State University protocol calculates the frequency for each stream monitoring locations using the following equation (U.S. Water Resources Council 1982):

$$\log(Q_{Tr}) = avg(\log(Q) + [K(Tr \times Cs)] \times \sigma logQ \qquad \text{[Equation 5]}$$

where Q_{Tr} is the discharge for the return period (m^3/s), Q is the peak flow for each year (m^3/s), Tr is the return period, and K is the frequency factor. K is a function of Cs, and the frequency factor table (U.S. Water Resources Council 1982). Cs, also known as the skew coefficient, is the statistical term describing the third central movement of the mean. Further, the Cs determines the shape of the flood frequency diagram.

To determine potential impacts of the flood, soil characteristics for each stream were collected from the Web Soil Survey (NRCS, 2019). The watershed area were quantified with the hydrology spatial data toolbox in ArcMap. Land use data were obtained from the National Agricultural Statistics Service using the CropScape- 2018 and 2019 cropland data layer. Using ArcMap, the 2018 land use and area for each watershed were calculated. Since the flood occurred in 2018 and I sampled in 2019, I investigated land use data to examine if there were any other major changes in the watersheds that could affect the nutrient spiraling experiment. The compared land use between 2018 and 2019 had no notable land use changes at the sampling locations. Land use was reclassified in ArcMap (ver. 10, ESRI, Redlands, Cal.) into five categories: grassland, urban, row crops, forest, water, or non-row crops.

INSTREAM SAMPLING

Over a two-day period at each site, I collected information on basic physicochemical parameters of stream sediment and water and then performed nutrient spiraling assays to assess nutrient cycling. As physicochemical sampling may have disturbed the stream bed, the nutrient spiraling assay was performed the following day.

At each stream, five pre-sampling locations and eight spiraling locations were measured, within a 100-300-meter stream reach. Sampled locations were determined using standard methods that are described briefly below (Jennifer L. Tank, Reisinger, and Rosi 2017).

At each of the five pre sampling locations, I assessed stream physicochemical parameters. In the thalweg at each location, stream temperature, dissolved oxygen concentrations, specific conductivity, and pH were determined using an YSI multimeter 556 (Yellow Springs Instruments, Yellow Springs, OH, USA). Stream discharge was determined using a Marsh-McBirney Flo-mate 2000 (Hach, Loveland, CO, USA).

Water samples were collected with a stream-rinsed, acid-washed 1L Nalgene bottles for analysis of total suspended solids (TSS), chlorophyll *a*, dissolved organic carbon, and nutrient concentrations, as described below.

Sediment samples were classified based on texture and used for a carbon and phosphorus analysis.

After collecting the stream water and sediment samples, I estimated nutrient spiraling metrics. To quantity the nutrients and conservative tracer salts needed for the injectate the following calculation was used:

$$Q_1 = Q * C_s / C_1 \qquad \text{[Equation 6]}$$

Where Q is stream discharge, C_s is the background nutrient concentration, Q_1 is the injectate drip rate into the stream, and C_1 is the nutrient concentration in the carboy. The salt quantity needed was calculated using the following equation with each nutrient:

$$N_g = Q * Ta * D * Cl * N_{NH4} \qquad \text{[Equation 7]}$$

Where N_g is grams of nutrient, Q is stream discharge (L/sec), Ta is Target addition (ug N /L), D is the inverse of the drip rate (min/L), Cl is liters in the carboy, and N_{NH4} is the N in ammonium, or P in phosphate. Conversions applied accordingly. The stream injectate was made using ammonium chloride (NH_4Cl), potassium phosphate ($K_2PO_4^{-2}$), and sodium bromide (NaBr). Target stream enrichments for ammonium (NH_4), phosphate (PO_4^{-2}) and bromide (Br) were 100 ug Br/L, 50 ug N/L, and 250 ug P/L respectively. I used the same nutrient enrichment ratios for each site to better compare the spiraling results across regions of the state.

A pump was placed adjacent to the stream with a steady drip of injectate of between 40-60 mL/min. After the stream became saturated with nutrients, stream water samples were collected at each of the eight spiraling locations. Basic water quality parameters from the YSI 556 multimeter were also taken at this time. Water samples were filtered either in the field or in the lab with a GC/F Whatman glass filter and put on ice until stored properly in the lab freezer.

LAB PROCEDURES

After each sampling event, stream water samples were returned to the lab and processed immediately. Unfiltered water samples were analyzed for total nitrogen (TN), total phosphorus (TP), TSS, and chlorophyll *a*. Filtered water samples were screened through a Whatman GC/F filter and the resulting water sample used for bromide (Br^-), nitrate (NO_3), ammonium (NH_4^+), total dissolved nitrogen (TDN), total dissolved phosphorus (TDP), soluble reactive phosphate (SRP), and dissolved organic carbon (DOC). Sediment samples were centrifuged and drained of excess water before freezing.

Total suspended solids and organic matter were calculated using a loss on ignition protocol. Briefly, filters were pre-combusted for 4 hours at 550°C, and then cooled in desiccator for a minimum of 24 hours. Filters were then weighed, filtered with sample water, and dried again for a minimum of 24 hours at 65°C. Then filters were weighed a second time, combusted a second time (550°C for 4 hours), and then weighed a final time to calculate TSS and Loss-on-ignition carbon (LOI-C) (Heiri, Lotter, and Lemcke 2001). Chlorophyll *a* and pheophytin concentrations on a Whatman GC/F filter were determined spectrophotometrically following a MgCO3-buffered methanol extraction (Steinman and Duhamel 2017; APHA 2005).

For each analyte from the nutrient spiraling assays and the physicochemical assay, one background and two plateau samples were analyzed. If background or plateau samples were not similar (similarity was defined as < 20% difference in replicate stream background values or site plateau values), another of the replicate samples was included in the analysis.TN and TP concentrations were analyzed using a persulfate digestion followed by a cadmium and an ascorbic acid reduction method on an Astoria Pacific autoanalyzer (APHA, 2005, Astoria Pacific, MA, USA).

NO_3 and Br concentrations were measured using an ion chromatography machine with a chemical suppression of eluent conductivity using a Dionex ICS-1100 (Thermofisher, Waltham MA, USA) (APHA, 2005). NH_4 concentrations were measured using the OPA method with fluorometry on an AquaFlour 9000-010 fluorometer (Taylor et al., 2007; Holmes et al., 2011). TDN and DOC concentrations were determined using the ASTM D8083-16 method on a TOC-L CPN Shimadzu (Shimadzu Corporation, 2017, Kyoto, Japan). SRP was analyzed using the molybdenum blue method and 2005 APHA standard methods for the examination of water and wastewater on a spectrophotometer (Genesys 150 UV-Vis Spec, Thermofisher, Waltham MA, USA).

Using the difference between TDP and SRP, dissolved organic phosphorus (DOP) was calculated. Particulate phosphorus (PP) and particulate nitrogen (PN) were calculated using the difference between total and total dissolved constituents for each nutrient. Dissolved organic nitrogen (DON) was calculated from the difference between TDN and DIN. Using the pre-sampling nitrogen, and phosphorus data, dissolved organic nitrogen (DON), particulate nitrogen (PN), dissolved organic phosphorus (DOP) and particulate phosphorus (PP) concentrations were calculated. Some of the calculations required for DOP, PP, DON, and PN, were negative due to the similar concentration values. Negative values of compounds indicated the results were smaller than the analytical error for the analysis. Sampled were field blanks analyzed for each compound to determine contamination.

Sediment samples were analyzed for texture and chemical content. To determine texture, sediment samples were sent to WARD Laboratories Inc. (Kearney, NE), for analysis by the hydrometer method. To determine sediment carbon concentrations, the loss on ignition procedure was used (Heiri, Lotter, and Lemcke 2001). To determine sediment phosphorus concentration, sediment samples used for LOI were then analyzed spectrophotometrically following the method of Andersen (1976).

CALCULATING NUTRIENT SPIRALING METRICS

Three main calculations are required to quantify nutrient spiraling: nutrient uptake length, uptake velocity and aerial uptake rate. To calculate the nutrient uptake length (S_w), an exponential decay model is used:

$$Ln\ N_x = Ln\ N_o - kx \qquad \text{[Equation 8]}$$

Where N_x is the background corrected plateau nutrient concentration saturation at *x* meters downstream of the injection point. N_o is the background corrected nutrient concentration at the site of injection, *k* is the exponential decay rate, and x is the x is the number of meters downstream. Nutrient uptake length (S_w) is calculated with the inverse of the decay rate:

$$Sw\ (m) = k^{-1} \qquad \text{[Equation 9]}$$

The uptake velocity (V_f) is the rate which nutrients move from the water to the stream relative to nutrient concentrations. Uptake velocity is calculated as:

$$Vf\ \left(\frac{m}{min}\right) = Qk/\mathrm{w} \qquad \text{[Equation 10]}$$

Where Q is discharge (m^3/min) and *w* is the mean stream wetted width (m). Aerial uptake rate (*U*), reflects the magnitude of the flux of inorganic element from the water to the biota. Areal uptake rate (*U*) is calculated as:

$$U\ (mg\ N\ m^{-2}\ min^{-1}) = Vf/Nb \qquad \text{[Equation 11]}$$

Where V_f is the uptake velocity and N_b is the background nutrient concentration prior to release.

STATISTICAL ANALYSIS

To compare impacts of the flood on stream sediment, physicochemical parameters, and nutrient uptake characteristics, I used a Shapiro-Wilk’s normality test in R to examine if my samples distribution. If my samples followed a normal distribution, I performed a paired T-test between 2018 and 2019 values using R (R 2017). If my samples did not follow a normal distribution (NO_3, NH_4, SRP, and DOP), I used a Wilcoxon signed rank test between the 2018 and 2019 values also using R. All background data (e.g., physicochemical parameters) are presented as means and standard deviations of the samples collected at each sampling location.

Some sampled stream locations had an extremely high uptake length (<3x the stream sampled reach), which implies the stream was saturated with this nutrient and would only be transported downstream instead of utilized. In cases of this I used 3x the sampled stream reach length as the nutrient uptake length and adjusted the vertical velocity and aerial uptake values to zero. With nutrients being transported downstream and not utilized, the V_f and U would be negligible.

RESULTS

COMPARING FLOOD MAGNITUDE AMONG SAMPLING LOCATIONS

LOST creek had the smallest return interval followed by the three Antelope streams (ACRR, ACUP, and ACEF). The NVP locations and SBML had the highest return intervals with 167 and 111 years respectively (Table 2). LOST had the most years of data, over 100 years, and the highest discharge values in March 2019. SBML and the NVP locations had the highest annual discharge in 2018.

Table 2 Spiraling locations with calculated return intervals for the 2019 flood along with the March discharge values, years of data and USGS unique ID name a number. The NVP and SBML had the highest return intervals and annual 2018 discharge.

Stream ID	Return Interval (years)	March 2019 Values of Q	Annual Q (2018)	Years of data	Location	USGS Unique ID
Antelopes (ACEF, ACRR, and ACUP)	11.83	182	8.55	8	Antelope Creek at 27th Street at Lincoln, NE	6803300
LOST	2.67	10500	111.20	101	Republican R NR Hardy NE	6853500
SBML	111.16	890	506.70	47	Middle Loup River at Dunning NE	6775500
NVP (4)	167.1	4720	201.90	21	Long Pine Creek near Riverview	6463500
CCSP	47.47	708	NA	19	Niobrara River near Hay Springs	6456500

SBML had the highest watershed area and NVP1 had the smallest watershed area. NVP and SBML locations were predominately grassland. The antelope locations had the highest coverage of urbanization. LOST had the highest coverage of row crops and CCSP had the highest coverage of non-row crops, such as oat, grapes, alfalfa, and hay (Table 3).

Table 3 Using the reclassification tool in ArcMap, land uses at spiraling sampling locations were determined.

2018 Watershed %	NVP-HZ	NVP1	NVP-MC	NVP-AH	ANT	CCSP	LOST	SBML
Grassland	95.20	92.24	57.10	91.61	5.51	49.67	38.84	90.56
Urban	1.37	1.64	2.84	0.18	90.94	1.36	4.57	0.59
Row Crops	NA	5.75	1.60	8.01	0.30	0.14	51.49	0.00
Forest	3.21	NA	25.51	NA	0.86	3.02	1.75	0.02
Non-row	0.01	NA	10.01	NA	0.31	44.99	2.68	0.09
Water	0.18	0.35	2.90	0.18	2.04	0.80	0.63	8.72

STREAM PHYSCIOCHEMICAL PARAMETERS

To determine the impact of the 2019 flood, stream characteristics in 2018 and in 2019 were compared. The average 2018 pre-flood sampling campaign had a relatively basic pH (8.61 ± 0.55), while the average 2019 post-flood locations had an acidic pH (8.07 ± 0.51) (T= 2.18, $p \leq 0.05$). Temperatures (21.84 ± 4.80, $p \geq 0.05$) and discharge (62.31 ± 58.38, $p \geq 0.05$) were higher post-flood and did not differ between years (Table 4). Although conductivity (613.62 ± 440, $p \geq 0.05$) was higher post flood it did not differ between sampling years. DO % was higher in the sampled streams before the flood but was not significant (93.30 ± 19.94, $p \geq 0.05$).

Table 4 The 2018 and 2019 YSI pre-sampling water quality measurements for each site including reach length, pH, temperature, conductivity, dissolved oxygen (DO%), and discharge.

Date	Reach ID	Reach Length (m)	pH	Temperature (°C)	Conductivity (uS cm^{-1})	DO %	Discharge (L/s)
180716	ACUP	270	9.53	32.3	435	119	148.5
180719	ACRR	210	8.28	28.26	625	93.64	19.2
180716	ACEF	250	9.74	32.76	427.16	136.72	104.64
180726	CCSP	280	8.344	18.56	406.46	81.18	57.02
180724	SBML	350	8.5	21.28	357.56	80.38	137.74
182909	LOST	200	8.646	10.72	863.4	68.38	0.81
180809	NVP AH	241	8.28	16.34	169.76	88.54	14.7
180808	NVP MC	280	8.392	16.6	401.98	87.16	66.75
180807	NVP HZ	200	8.136	16.98	149.74	87.88	27.4
180731	NVP1	200	8.272	15.6	179.52	90.14	15.13
Date	Reach ID	Reach Length (m)	pH	Temperature (°C)	Conductivity (uS cm^{-1})	DO %	Discharge (L/s)
190703	ACUP	270	7.674	24.08	536.06	70.62	198.99
190702	ACRR	210	7.878	24.42	595.74	61.5	82.61
190703	ACEF	200	8.588	27.44	463.3	110.5	28.19
190716	CCSP	280	8.178	16.908	666.6	83.9	12.13
190723	SBML	350	8.96	26.028	626.4	87.36	64.14
190820	LOST	200	8.528	28.484	1787	91.51	66.18
190814	NVP AH	241	7.5	17.7	283.6	89.84	14
190814	NVP MC	280	7.694	19.786	582.2	90.7	110
190813	NVP HZ	200	NA	18.712	421.66	84.98	34.78
190813	NVP1	200	7.664	14.936	173.72	96.56	12.12

Stream water nutrient concentrations and sediment characteristics varied somewhat across the state. In 2018, two of the sites in the Niobrara Valley Preserve, NVP1 and NVP-MC, had the highest NO_3 concentrations (2.25 mg N L^-

^{1} ± 0.01 and 1.72 mg N L^{-1} ± 0.65, respectively). In 2018, ACEF had the highest concentration of NH_4 (1.00 mg N L^{-1} ± 0.07) and LOST had the highest concentration of SRP (2.01 mg P L^{-1} ± 0.23). In 2019 LOST had the highest NO_3 concentrations (4.28 mg N L^{-1} ± 0.23). Every stream in 2019 had NH_4 concentrations less than 1 mg/L (0.01 mg N L^{-1} ± 0.01).

In 2018 the average LOI C was 9.09 mg C L^{-1} ± 9.00. The average TSS was 52.45 mg L^{-1} ± 34.89, the average chlorophyll concentration was 0.0 mg L^{-1} ± 0.12. The average pheophytin concentration was 0.00 mg L^{-1} ± 0.00, and the average DOC was 2.97 mg C L^{-1} ± 3.92. The average TDN concentration was 0.88 mg N L^{-1} ± 0.45, and the average TDP was 0.09 mg P L^{-1} ± 0.04. The average TN was 0.91 mg N L^{-1} ± 0.33, and the average TP was 0.09 mg P L^{-1} ± 0.04. The average PN was 0.06 mg N L^{-1} ± 0.16 while the average PP was 0.04 mg P L^{-1} ± 0.04. The average DON was 0.28 mg N L^{-1} ± 0.33 while the average DOP was 0.59 mg P L^{-1} ± 0.44.

In 2019 the average LOI C was 19.19 mg C L^{-1} ± 27.62. The average TSS was 69.04 mg L^{-1} ± 78.27. The average chlorophyll concentration was 0.01 mg L^{-1} ± 0.04, and the average pheophytin concentration was 0.22 mg L^{-1} ± 0.48. The average DOC was 5.13 mg C L^{-1} ± 3.50. The average TDN concentration was 0.75 mg N L^{-1} ± 0.52, and the average TDP was 0.01 mg P L^{-1} ± 0.05. The average TN concentration was 0.51 mg N L^{-1} ± 0.52 while the average TP was 0.10 mg P L^{-1} ± 0.11. The average PN was 0.15 mg N L^{-1} ± 0.25 while the average PP was 0.02 mg P L^{-1} ± 0.06. The average DON was 0.24 mg N L^{-1} ± 0.28 while the average DOP was 0.53 mg P L^{-1} ± 0.36.

Sediment at the Niobrara sampling locations and SBML had a sandy soil texture. Sediment at the remaining sites, CCSP, LOST, and the Antelope streams had sandy loam, loam, and sandy clay loam textures, respectively. In 2018 there was a higher percentage of sand (82.90 % ± 14.60), while in 2019 there was a higher percentage of silt (11.44 % ± 13.10) and clay (9.33 % ± 7.91) at the sampled locations. There was more carbon and phosphorus in the sediment in 2018 (0.01 mg C L^{-1} ± 0.01, and 300 mg P L^{-1} ± 302) when compared to 2019 (0.01 mg C L^{-1} ± 0.02, and 452 mg P L^{-1} ± 456).

Most of the water chemistry did not differ between 2018 and 2019. Most of the nitrogen species (TDN: T = 0.89, $p \geq 0.05$, DON: T =0.27, $p \geq 0.05$, and PN: T = -1.90, $p \geq 0.05$) and most of the phosphorus species (TP: T = 0.64, $p \geq 0.05$, DOP: T =0.32, $p \geq 0.05$), and PP: T= 0.27, $p \geq 0.05$) did not very pre- and post-flood.

Nitrate (T= -6.15, $p \leq 0.05$), ammonium (T= 3.33, $p \leq 0.05$), TN (T= 4.74, $p \leq 0.05$), soluble reactive phosphorus (T = 1.29, $p \leq 0.05$), and TDP (T = 8.53, $p \leq 0.05$) varied. NH_4 decreased by 11%, NO_3 increased by 171%, TN decreased by 43%, SRP decreased by 26%, and TDP decreased by 95%.

Similarly, TSS (T = -1.25, $p \geq 0.05$), and Chlorophyll a (T = 1.87, $p \geq 0.05$), did not differ between sampling years. However, DOC and LOI C did differ between sampling years, both DOC (T = -2.85, $p \leq 0.05$), and LOI C (T= -2.39, *p*

$\leq$0.05). Pheophytin (T= -3.27, $p \leq 0.05$) was also significant between 2018 and 2019 (Figures 19 through 24, respectively).

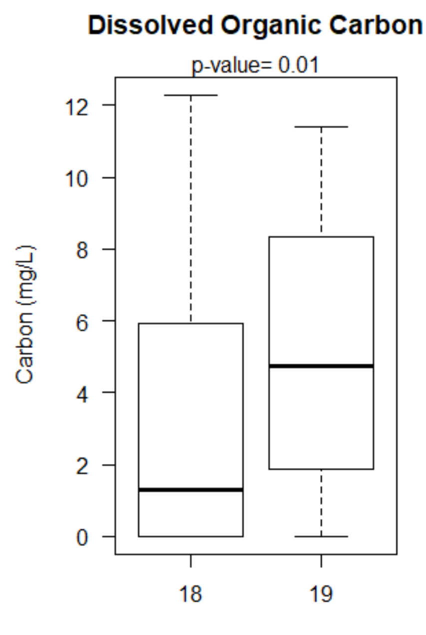

Figure 19 DOC concentrations from 2018 and 2019.

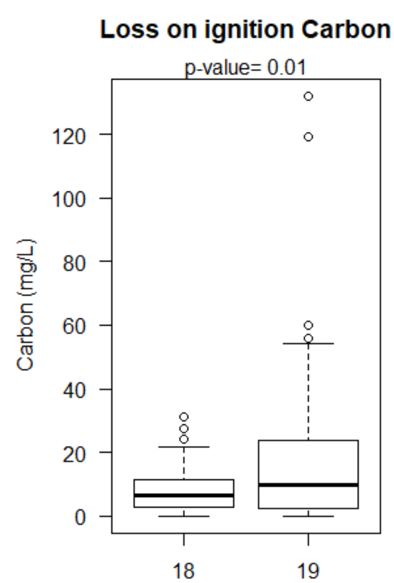

Figure 20 Loss on ignition carbon concentrations from 2018 to 2019.

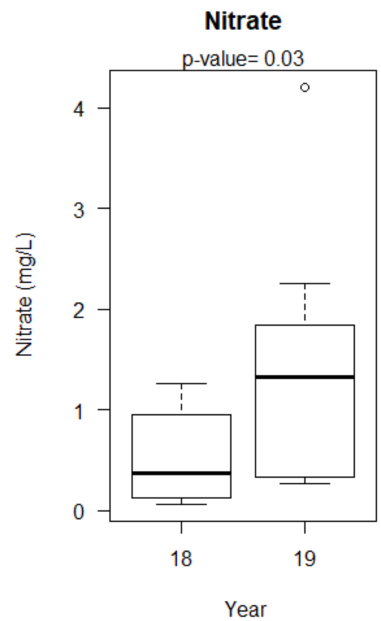

Figure 21 Nitrate concentrations from 2018 and 2019

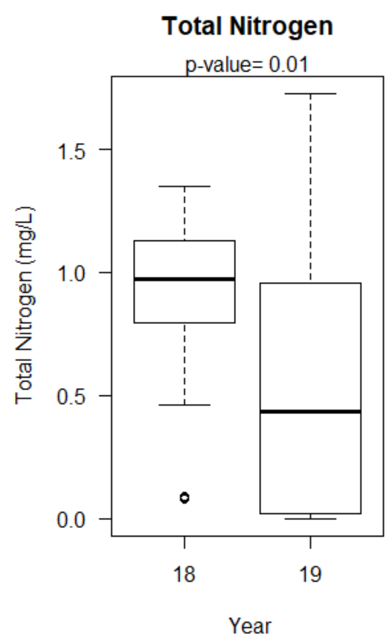

Figure 22 Total nitrogen concentrations from 2018 to 2019.

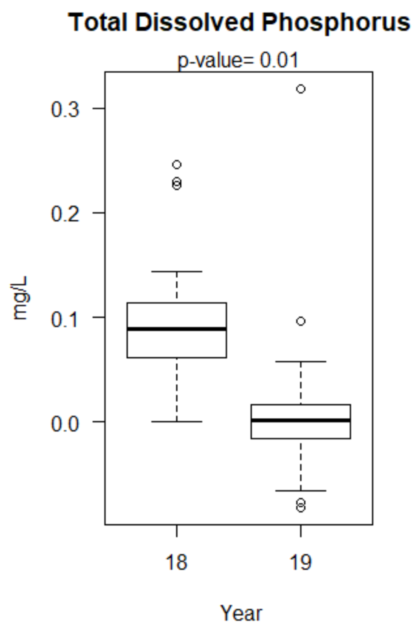

Figure 23 Total dissolved phosphorus concentrations form 2018 to 2019.

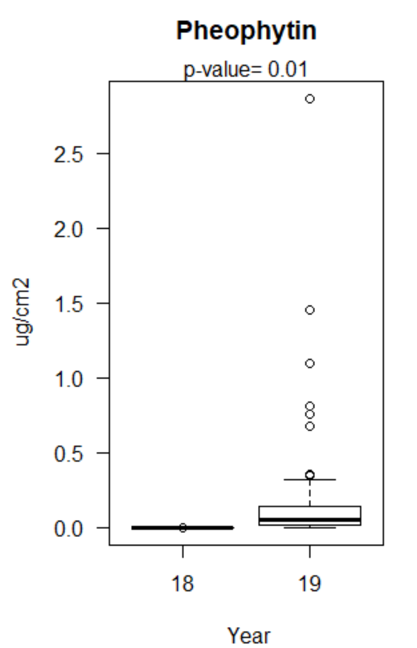

Figure 24 Pheophytin concentrations from 2018 to 2019.

Similarly, sediment characteristics did not differ between 2018 and 2019. The differences between percent sand (T= 0.46, $p \geq 0.05$), silt (T= -0.68, $p \geq 0.05$), and clay (T= -0.08, $p \geq 0.05$) was not significant between sampling years. Carbon (T= -1.81 $p \geq 0.05$) and phosphorus (T=-1.78, $p \geq 0.05$) within the stream bed sediment did not differ.

RESULTS
SPIRALING CHARACTERISCTICS

UPTAKE LENGTH

In 2018 the average uptake length (Sw) of NH_4 was 631 m ± 348, with a range of 167 m in LOST, to 12,000 m in SBML. In 2018 I removed four locations (ACEF, ACRR, NVP1, and NVP-AH) from the analysis for NH_4 uptake length due to laboratory analytical error. For the SRP uptake length I also had an extremely high average with a range of 102 m in ACUP and <16,000 m in ACUP. In 2018 the mean S_w of PO_4 was and 500 m ± 260. Only three locations could be calculated for other spiraling metrics due to the high uptake length (ACUP, ACRR, and SBML).

In 2019 there was a lower average of NH_4 and PO_4 uptake lengths. The mean Sw of NH_4 and PO_4 was 463 m ± 349 and 375 m ± 352. NVP1 had the smallest uptake length for NH_4 (89m) and PO_4 (85m), while SBML had the highest (<12,000m) NH_4 and PO_4 uptake length.

There were three locations with high uptake values in 2018, that impacted the other spiraling metrics to be calculated for three sites for NH_4 (ACUP, SBML and NVP-MC), and five for the PO_4 analysis (ACEF, CCSP, NVP1, NVP-HZ, NVP-MC, and NVP-AH). In 2019 the lowest NH_4 uptake length was 89 m at NVP1 and the highest length was over 21,000m at SBML.

There were five locations with high uptake values in 2019, that impacted the other spiraling metrics to be calculated for NH_4 (SBML, NVP-HZ, NVP-MC, NVP-AH, and LOST) and two sites for PO_4 (SBML and NVP-AH).Although uptake length varied from sampling years there is not a statistical difference in NH_4 (T= 0.93, $p \geq 0.05$) uptake or PO_4 uptake length (T= 0.88, $p \geq 0.05$) (Figure 25 and Figure 26).

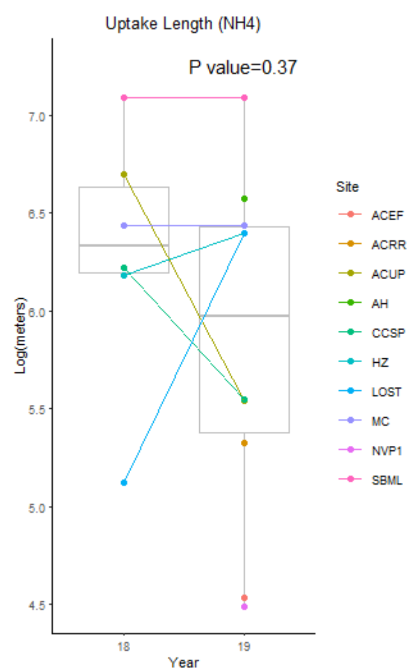

Figure 25 Uptake length of ammonium from 2018 to 2019.

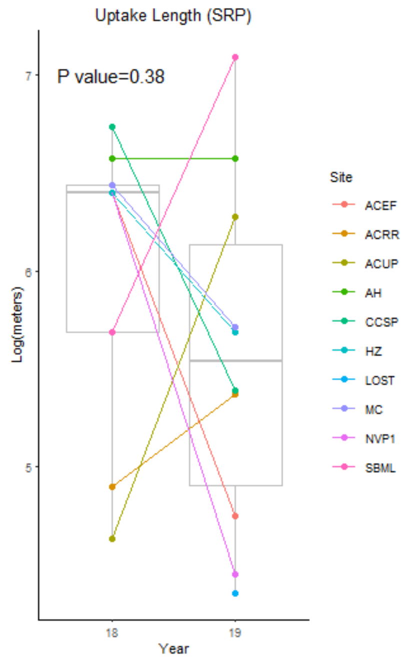

Figure 26 Uptake length of phosphorus from 2018 to 2019.

VERTICAL VELOCITY

Of the 10 locations sampled in 2018, the uptake length in only three of the sites were detectable (less than 3X the sampled reach) for NH_4 and in only six of the sites for P- the median V_f for NH_4 was 0.25 mm/min ± 0.42 and 0.42 mm/min ± 0.42 for PO_4. CCSP had the highest V_f as 0.85 mm/min and LOST had the lowest as 0.00 mm/min.

In 2019, five sites had detectable uptake lengths NH_4 and two sites had detectable uptake lengths for P. There was a varied median of NH_4 and PO_4 vertical velocity at 0.20 mm/min ± 0.42 for NH_4 and 0.42 mm/min ± 0.90 for PO_4. V_f for NH4 ranged from 0.09 mm/min at NVP1 to 1.38 mm/min at ACUP. V_f for PO4 ranged from 0.00 mm/min at NVP-AH and SBML to 2.88 mm/min at ACUP. The vertical velocity did not differ from sampling year of NH_4 (T= 0.08,

$p \geq 0.05$) (Figure 27) or PO_4 (T= 0.75, $p \geq 0.05$) (Figure 28), but the number of streams that changed from non-detectable to detectable was two for NH_4 and four for PO_4.

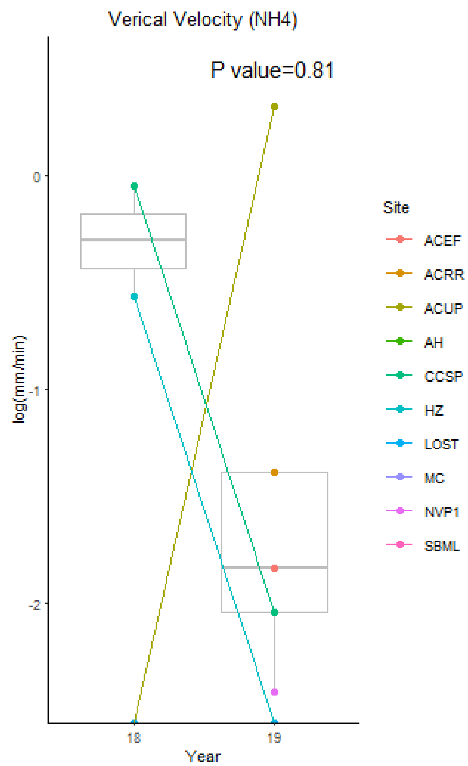

Figure 27 Vertical velocity of ammonium from 2018 to 2019.

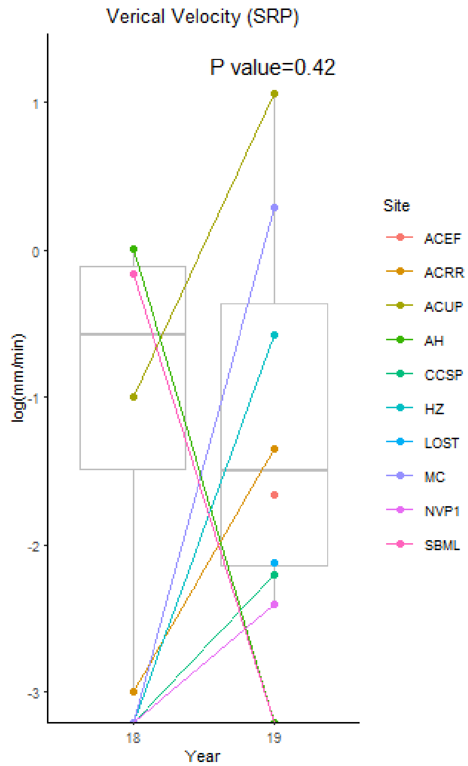

Figure 28Vertical velocity of phosphorus from 2018 to 2019.

AERIAL UPTAKE RATE

In 2018, when detectable, the average aerial uptake rate for NH_4 of 9.28 ug m^2/min ± 0.30. Aerial uptake rates varied over several orders of magnitude, i.e., CCSP had the highest rate of U_{NH4} at 28.46 ug m^2/min and LOST with the lowest rate of 0.10 ug m^2/min. The aerial uptake rate for PO_4 was low with an average rate of 1.11 ug m^2/min ± 2.61. ACUP had the lowest value for U_P of 0.01 ug m^2/min and NVP-AH had the highest vales of 7.83 ug m^2/min.

In 2019 the average NH_4 and PO_4 aerial uptake rate at 8.58 ug m^2/min ± 0.05 for NH_4 and 19.59 ug m^2/min ± 58.61 for PO_4, but these differences were not significant from 2018 (U_{NH4} T= 0.08, $p \geq 0.05$; U_P = -0.99, $p \geq 0.05$). U_{NH4} ranged from 0.03 ug m^2/min at ACUP to 52.89 ug m^2/min at ACRR. U_P ranged from 0.02 ug m^2/min at ACUP to 186.38 ug m^2/min at NVP-MC (Figure 29 and Figure 30).

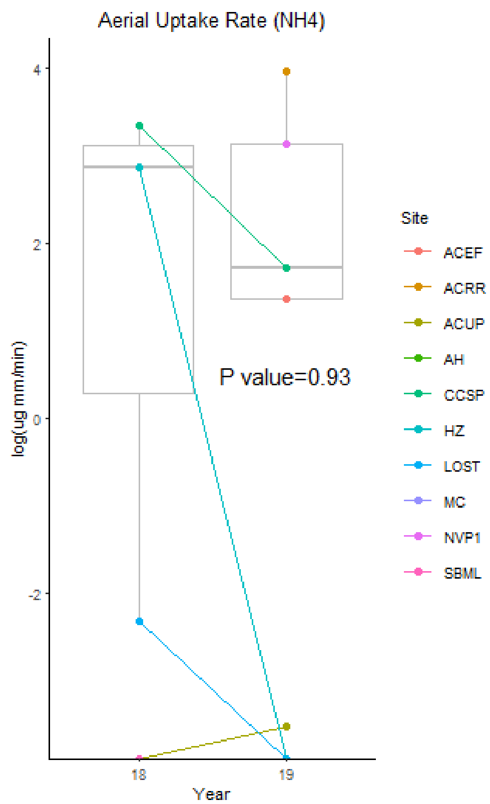

Figure 29 Aerial uptake rate of ammonium from 2018 to 2019.

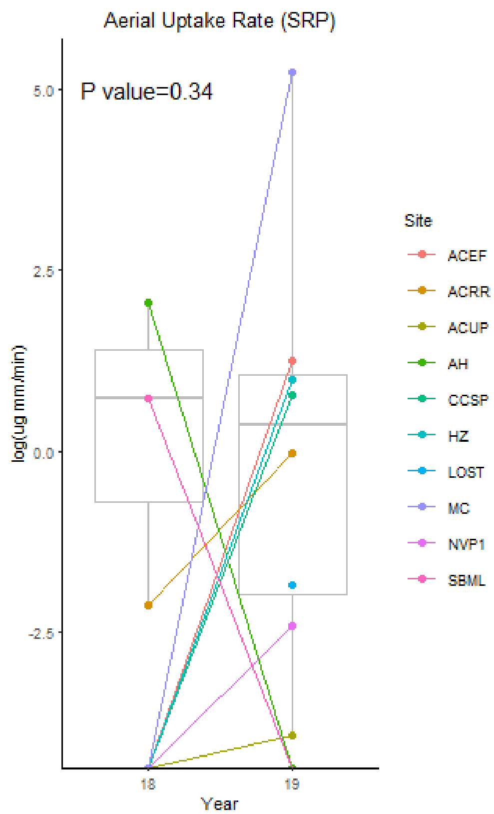

Figure 30 Aerial uptake rate of phosphorus from 2018 to 2019.

To examine the impacts of flood magnitude on nutrient spiraling parameters, I plotted the return period values against the change of aerial uptake of NH_4 and PO_4. There was no correlation (Figure 31 and 32).

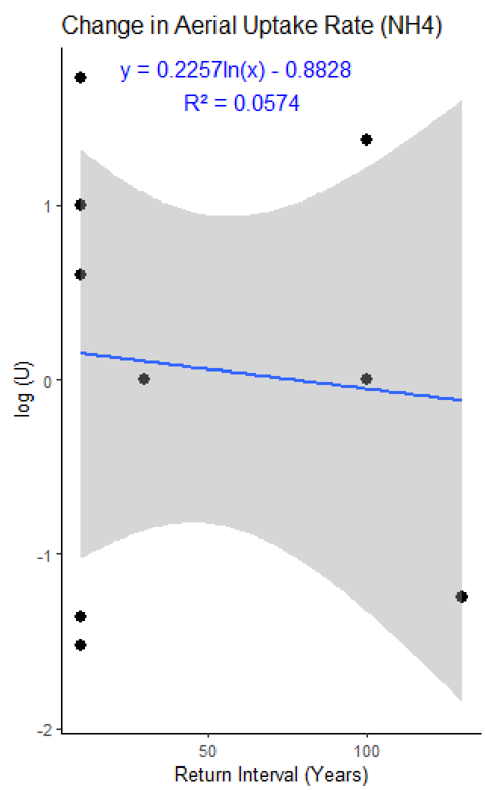

Figure 31 Change in aerial uptake rate of ammonium compared to flood return periods.

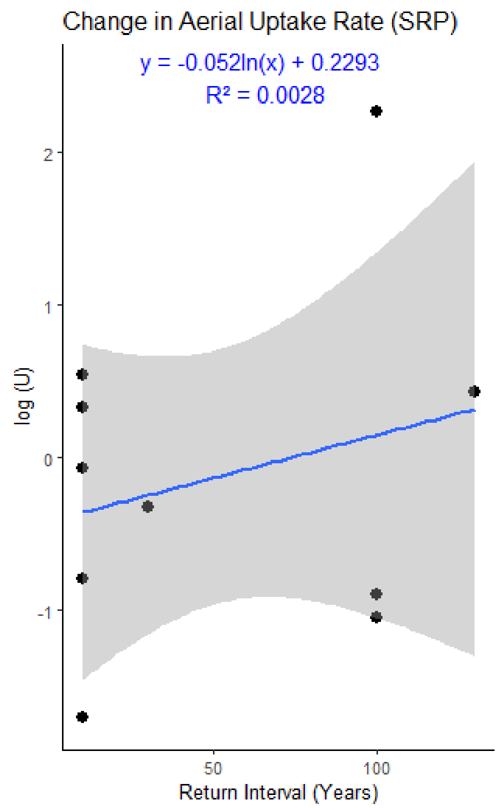

Figure 32 Change in aerial uptake rate of phosphorus compared to flood return periods.

DISCUSSION

COMPARING THE FLOOD BETWEEN SITES

Longer, more complete sets of data give more accurate determinations of return periods. LOST has more discharge data (101 years) compared to any other site. This amount of data could influence the return interval value, which was 2.67, either increasing or decreasing depending on changes in the catchment. Ten or more years of data is typically required to perform a frequency analysis for the determination of return periods. Having 30 years of data, compared to 10 years of data provides more confidence on the results. The maximum discharge values for LOST in 2019 were only the 19th highest in the entire dataset suggesting LOST had had floods higher in magnitude in previous years.

CCSP is located in a dense vegetation and forest covered state park, which could have affected the return period. Dense vegetation and forest cover increase infiltration rates, lowering runoff and slowing stream flow from precipitation events. CCSP had the highest elevation of all sites with the most land use coverage of forest. This could have helped minimize the flooding effect on the return period giving CCSP a return period of 47 years for 2019.

The four NVP streams are within the Northwestern Great Plains ecoregion, which is a semiarid rolling plain of shale and sandstone with sporadic buttes. Agriculture practices are restricted by erratic precipitation and lack of irrigation. The four NVP streams had the highest return value of 167 years. NVP also had high average March discharge values, likely due to the high precipitation and rapid temperature flux, enabling snow melt to runoff into the streams and rivers. Over 70% of the water in the Niobrara River comes from groundwater from the Ogallala, Aquifer. The large drainage area of the Niobrara River and with the majority of the soil being sand, this could have helped influence a high discharge and return period.

The Sandhill's contains numerous lakes and wetlands without stream connectivity (Primary, 1999). This lack on connectivity could influence floodwater translocation. The SBML had one of the highest return intervals and the highest annual Q value. The high annual Q values would have a high influence on the flood return period in SBML, which was 111 years.

The three Antelope sites are within the Western Corn Belt Plains ecoregion. More than ¾ of the Western Corn Belt Plains is used for cropland agriculture with the majority of the remaining land used as forage for livestock. Agricultural practices have been shown to have negative effects on water quality and water infiltration rates. The effects of urban or agriculture development on peak flows is generally much greater for low return interval floods. Typically, during larger floods, the soil is saturated lowering the capacity to absorb additional rainfall. Under these conditions the precipitation runoffs paved or saturated surfaces and becomes streamflow.

Two of the three Antelope streams were channelized (ACUP and ACEF) to help control water. These streams were downstream of a reservoir and a weir to help controlled flow, thus lowering the return intervals. ACUP was designed to be flood control by the U.S. Army Corps of Engineers and the Lower Platte South Natural Resources District in 2012 (Laukaitis, 2011). The flood plain control at ACUP was part of the Antelope Valley Project, aimed at controlling water flow and improving water quality, a task which costs 4.75 million dollars (Union Plaza Grand Opening, 2012). If there are significant changes in the flow patterns, by an impoundment or diversion of flow, the return period for the annual peak streamflow at a given location will change. This, and the minimum years of discharge data, could have affected the return interval values, which all three Antelope sites had under a 12-year return period.

PYSCOIOCHEMICAL PARAMETERS

How did stream differ between 2018 and 2019?

Between 2018 and 2019 various stream physicochemical parameters changed such as pH, LOI C, DOC, and pheophytin. Of the nitrogen species, nitrate and ammonium, varied and of the phosphorus species, SRP and TDP changed.

Did the flood influence these changes?

Within aquatic ecosystems, the concentrations of DOC is more related to climate and landscape topography when compared to internal properties and processes of aquatic ecosystems (P J Mulholland 2003). Both carbon species (DOC and LOI C) increased after the flood as well as pheophytin. Low nutrient levels may prohibit the remineralization of DOC in oligotrophic waters, (Dittmar and Stubbins 2013) but these stream were saturated with nutrients suggesting that remineralization of DOC could have increased carbon concentrations as well as pheophytin. Because phaeophytin is one of the degradation products of chlorophyll, the ratio of chlorophyll *a* to phaeophytin can serve as an indicator of the physiological condition of phytoplankton within samples. Since our phytoplankton concentrations increased this could mean that there was increase biological activity after the flood.

Carbon is often in excess amounts relative to macronutrients such as N or P in photoautotrophs (Hessen and Anderson 2008). The higher concentrations of carbon within the sampled streams in 2019 could have influenced the dissolved organic nitrogen concentrations. DON is a subset of the DOC pool that also contains N (Sipler and Bronk 2015). Further, this would enable the TN concentrations to be affected. Dissolved organic nitrogen has the largest quantity of total dissolved nitrogen in rivers, streams, and estuaries.

Temperature and pH determines whether ammonia or ammonium is present. High pH and low temperatures, convert ammonium to ammonia. Ammonia is a highly soluble, biologically active compound found in most waters as a normal biological degradation product of nitrogenous organic matter. A common source of ammonia is discharged industrial waste that contain ammonia and fertilizer. Pre-flood sampling had lower ammonium concentrations but higher nitrate concentrations. When compared to nitrate and phosphorus, ammonium concentrations decrease more quickly when moving downstream (Tank et al., 2017b). Retention variability of ammonium is influenced predominantly by biogeochemical reactivity controls (Marcé et al. 2018),

Biological activity and nutrient concentrations would influence U, V_f and S_w. Due to differences in the energetic cost of assimilation, ammonium is generally preferred over nitrate as an inorganic N source, but the rate of nitrification is affected by the natural concentration of ammonium (Leoni et al. 2018). Since ammonium concentrations decreased biotic assimilation could have been happening. Biotic assimilation is the main pathway for nitrate removal (Hall et al. 2013), yet the start of nitrification can vary as a function of trophic status (Leoni et al. 2018). Differences between nitrate and ammonium uptake kinetics are expected (Ribot et al. 2013), due to nitrogen transformations having the ability to be carried out by various organisms (Bothe and Drake 2007). Longer travel times can create greater processing within the stream and our sampling locations varied in reach length thus enabling a varied uptake.

Phosphorus exists in a many forms in natural waters, in both dissolved or particulate phases, and as inorganic or organic (Jarvie, Withers, and Neal 2002). Although the phosphorus cycle is better understood when compared to nitrogen, phosphorus is often limiting in natural waters because it has no gaseous phase, so it is not accessible. Unlike nitrogen, phosphorus has poor mobility due to sorption kinetics of dissolved forms. This means phosphorus will likely stay in local environments (Thomas and Cebrian 2008) unless removed by flooding's or dredging.

In our study, TDP and SRP decreased from 2018 to 2019, suggesting P removal during sampling years. The higher pH in 2018 could have influenced phosphorus utilization by periphyton photosynthesis. Periphyton photosynthesis can locally increase pH, which can lead to increased precipitation of calcium phosphate, and long-term burial of P (Dodds 2003).

River networks can become saturated with nutrients under higher flow conditions because supplies to the system increase faster than sink processes. With an increasing supply, the saturated river network scales become limited by the previously unmet demand in downstream aquatic ecosystems (Wollheim et al. 2018).

SPIRALING

Streams uptake parameters varied considerably across sites in both 2018 and 2019. According to a 2006 nutrient spiraling review the interquartile range in

Sw in a first order stream is 23 to 275 m for NH_4, and 24 to 161 m for PO_4 (Ensign and Doyle 2006). All of our stream locations were first order streams but our S_w varied. S_w increases more slowly than does distance downstream from headwaters (Hall et al. 2013), and half of our streams were headwater streams.

The interquartile range of V_f reported by Ensign and Doyle (2005) is 2.5 to 17.0 mm/min for NH_4, and 1.5 to 6.6 mm/min for PO_4. Our ranges for NH_4 and PO_4 V_f were on the lower side of the range compared to Ensign and Doyle (2005). Vf increases across the range of possible values previously observed for different constituents, the capacity of the network to remove constituents increases considerably (Talbot et al. 2018).

Aerial uptake rate (U) does not have a direct correlation with discharge or water velocity of a stream, enabling it to be compared with various stream ecosystems (Weathers et al. 2013). The interquartile range of U in a first order stream is 5.3 to 52.6 mg m^2 /min for NH_4, and 3.4 to 15.7 mg m^2 /min 1 for PO_4 (Ensign and Doyle 2006). Most of the sampled streams were also on the order range for U_{NH4} and U_{PO4}.

There was no statistical difference in NH_4 or PO_4 spiraling metrics from 2018 to 2019. This suggests either some resilience in our sampled locations or after sampling the streams up to seven months gave enough time for the streams to recover. Biological communities in river systems can respond to disturbances faster than those of other aquatic systems (Sabater 2008).

The high uptake lengths and low uptake velocity and areal uptake values, could also suggest that these small streams were enriched with N and P well beyond their biological demand for these nutrients.

SPECIFIC SITES

CCSP is located in Chadron State Park, a public reactional area located in Nebraska National Forest. CCSP was more populated than any other site with organisms of many trophic levels, which likely increased/decreased nutrient uptake, as the fate of chemical elements are strongly linked by biotic demand and uptake (Welti et al. 2017; García et al. 2017). CCSP is a forested stream that experiences distant seasonal changes compared to other sites, therefore heterotrophic organisms are likely the dominant mechanism controlling nutrient (Patrick J Mulholland and Webster 2010).

ACRR is located in a dog park downstream from Holmes Lake, a popular recreational reservoir located in southeast Lincoln. ACRR has incised stream banks with little vegetation. Dogs frequently ran in and out of the stream during sampling. Disturbances within sampling from erosion or dogs could have influenced the larger uptake rate.

ACEF is a drainage ditch that is lined with riprap along the sides and Pre-cast Concrete Block Mat on the floor. Riprap maintains artificial habitats designed to navigate channels, stabilize banks (Hepp and Santos 2009), and increase surface area. Sampled water collected water was notably green during collection, suggesting biotic activity. Channel geomorphology and residence time have an indirect effect on nutrient uptake from the water column by dictating the exposure of water to the substrates (Ensign and Doyle 2006). The morphology of the ACEF could have decreased nutrient uptake. Nitrate uptake rates are significantly lower in channelized reaches compared to other less-disturbed ones (Kunz et al. 2016; Jones et al. 2018).

ACUP is a channelized drainage ditch. It is located downstream from a weir, which is a low dam built across a river to control water flow. Consistent low flow enables algae growth within the stream, attracting ducks and birds. Animals are important in nutrient cycling, possessing comparable input rates to major nutrient sources (Vanni 2002). Consumers play a direct role in steam nutrient cycling by excreting dissolved nutrients back into the food web that were previously unavailable (Small, Helton, and Kazanci 2009). Nutrient spiraling metrics in ACUP may have been impacted by animals.

The NVP and SBML locations had the highest return intervals, thus having a higher flooding impact. Both streams are located in a predominately sandy location. Sand has a higher hydraulic conductivity, meaning water can infiltrated faster. Barren locations within these sites could be more susceptible to flooding impacts. Although much of Nebraska was impacted by the flood, not every location was impacted equally. More research should be done on why these locations had high return intervals.

LIMITATIONS AND FUTURE WORK

Not every stream was impacted the same by this flood and floods can leave heterogeneous patchy legacies from the alterations in severity and influence ecosystem components differently (Parsons et al. 2006), those I may not have sampled the reaches most impacted by the 2019 flood. Sampling campaigns in this study occurred 5-7 months after the flood, which may have allowed time for ecosystems to rebuild and replenish. Future studies should examine flood effects on ecosystem processes by sampling directly after a flood. Nutrient limitation can differ across biological and biogeochemical processes occurring in various ecosystem scales and can vary considerably over time and space (Tromboni et al. 2018). Incorporating a nutrient diffusion substrata sampling campaign could have given a better understanding if a limited nutrient influenced the spiraling campaign.

CONCLUSION

In this study, I show that the relative demand for NH_4 and PO_4 varied across streams and suggest the sampled streams may have been resilient to the 2019 floods.

CHAPTER 4. CONCLUSION

The objectives of my thesis were to answer the questions: what was the extent of the 2019 flood in Nebraska?, and how did it impact stream ecosystem processes? I evaluated these questions by conducting two projects, one by exploring the current flooding technology and a second field project in ten streams across Nebraska.

In Chapter 2, I explored how return intervals from USGS stream gage stations showed the magnitude of the flood while also evaluating some factors that may have caused the flood such as climate, land use, and soil hydrological data. By analyzing the discharge records of 94 streams across the state, I analyzed flood stage records and calculated flood return intervals. While the flood stage was reached primarily along streams in the Niobrara, Platte, and Elkhorn River watersheds, seventeen streams across the state had a flood return interval of over 100 years. The average return interval was 48 years +/- 73. In a case study of two watersheds in Nebraska I found that infrastructure, natural stream regimes, and errors in stream gage monitoring stations contributed to the variability of within the two watersheds. This result was expected and further reinforced the need for continued communication between communities and state and federal unities.

In Chapter 3, I sought to examine how nutrient cycling in streams across the state may have been impacted by the flood. I used a nutrient spiraling protocol in 10 streams pre and post flood. The sampling stream locations varied from Chadron, Sandhill's, Niobrara, Superior and Lincoln Nebraska. The average return interval was 48 years ± 73. I also compared nutrient spiraling metrics in ten streams across the state in the summer before and after the flood to compare the flood impacts on stream ecosystem processes. In 2018 the mean uptake length (S_w) of NH_4 and PO_4 was 563m ± 348 and 600 m ± 260. The vertical velocity (V_f) in 2018 for NH_4 and PO_4 was 0 mm/min ± 0.42 and 0.02 mm/min ± 0.42 while the aerial uptake rate (U) mean was 0.03 ug mm/min ± 0.30 and 0 ug mm/min ± 2.61 respectively. In 2019 the mean S_w of NH_4 and PO_4 was 248 m ± 349 and 256 m ± 352. The V_f in 2019 for NH_4 and PO_4 was 0.04 mm/min ± 0.42 and 0.15 mm/min ± 0.90 while the U mean was 0.01 ± 17.16 ug mm/min and 0.57 ug mm/min ± 58.61 respectively. Although these streams varied in the magnitude of the 2019 flood, I showed that Nebraska streams are somewhat resilient from the flood. There were no differences ($p > 0.05$) of NH_4 or PO_4 in Sw, Vf or U between years. Yet, given the high uptake lengths and low uptake velocity and areal uptake values, these data also suggest that small streams are enriched with N and P well beyond their biological demand for these nutrients. Furthering our knowledge of the quantification of floods and their impacts on stream ecosystem processes can help mitigate the deleterious consequences of floods.

With the in increasing frequency and duration of floods becoming more prevalent, knowing how to mitigate floods and how these major floods may impact stream ecosystem processes is helpful to protect our natural resources. Results of both of these studies will provide helpful insight to this and improve ecological understanding of natural disasters.

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